A Review of the Potential Influence of Heavy Metals on Salmonid Fishes in the Campbell River, Vancouver Island, B.C.

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by

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ABSTRACT

Alderdice, D. F. and W. E. McLean. 1982. A review of the potential influence of heavy metals on salmonid fishes in the Campbell River, Vancouver Island, British Columbia. Can. Tech. Rep. Fish. Aquat. Sci. 1104: vii + 60 p.

Potential joint toxic effects of dissolved and extractable zinc, copper and cadmium levels in water samples taken from the Campbell River, Vancouver Island, B.C. over the past 12 years (1971-1982) are assessed for three salmonids native to the river--the chinook (Oncorhynchus tshawytscha) and coho (O. kisutch) salmon and the steelhead trout (Salmo gairdneri). As little direct evidence is available regarding toxicity of Campbell River water, the assessment is based on relevant data from the current toxicological literature. There has been a consistent increase in zinc levels in river water, rising from about 0.007 mg/L early in 1971 to about 0.048 mg/L at the end of 1980. No consistent rise in copper concentrations could be demonstrated in the same period. However at a mean water hardness of 21.5(as mg/L $CaCO_3$)(1971-1982) mean metal levels in the river in recent years have been 0.0464 mg/L of zinc (1981-1982), 0.0021 mg/L of copper (1980-1982) while cadmium has been below detection limits since 1971 (<0.0005 mg/L). In the last few years (1980-1981) "spikes" of high concentration have reached maxima of 0.066 mg/L zinc and 0.0035 mg/L copper. Since mid-1981 there is a suggestion that metal levels may have declined somewhat, a possibility requiring further monitoring of the water to substantiate.

Estimates of 96-h LC50 concentrations and data relating to maximum acceptable toxicant concentrations (MATC) are marshalled from the literature for the species examined in tests at water hardness levels near that of Campbell River water. From these data MATC values are estimated and compared with existing metal levels from the water samples. Assuming additivity of toxic effects (Zn+Cu), the number of toxic units (TU) equivalent to (Zn+Cu) concentrations in the samples are derived for the three species considered. Based on average heavy metal loads, Campbell River water is calculated to have been toxic to naive juvenile S. gairdneri, (those moving into the main river from feeder streams or from the Quinsam hatchery) for most of the period between February 1980 and September 1981. Pre-exposed (non-naive), juvenile salmonids incubated as eggs and hatched in the river are judged to be more resistant to heavy metals, presumably through induction of metallothionein proteins that tend to bind and detoxify heavy metal ions. For pre-exposed fish, Campbell River water appears still to be non-toxic although the extent of risk can not be quantified at this time. The Zn+Cu loads in the Campbell River appear to have become critical to naive juveniles within the last two years (1980-1981). In that period, levels of heavy metals in water samples are predicted to have been toxic as follows: 3 of 38 samples (chinook), 30 of 40 samples (S. gairdneri), and 0 of 38 samples (coho). In general, susceptibility of the three species to heavy metal toxicity is estimated as S. gairdneri >chinook >coho. Although there are differences in susceptibility of naive (feeder stream and hatchery) and pre-exposed (main Campbell River)

fish, small differences in water characteristics resulting from changes in upstream conditions (e.g. decreases in hardness or pH) could increase the toxic potential of current heavy metal loads in the river water. It is concluded that water quality in the Campbell River has reached a critical state with regard to the well-being of salmonids native to the river. In view of the apparent severity of the current situation, the following recommendations are made: (1) immediate remedial action should be taken to protect indigenous salmonids in the Campbell River by reducing zinc and copper concentrations now occurring in the river, and (2) an investigation should be instituted to determine maximum acceptable heavy metal levels (Zn+Cu+Cd) for naive and pre-exposed native salmonids as well as representative organisms (algae, invertebrates) in the supporting food web in the Campbell River system.

RESUME

Alderdice, D. F., and W. E. McLean. 1982. A review of the potential influence of heavy metals on salmonid fishes in the Campbell River, Vancouver Island, B.C. Can. Tech. Rep. Fish. Aquat. Sci.

On évalue, pour trois espèces de salmonidés indigènes, c'est-à-dire le saumon quinnat (Oncorhynchus tshawytscha), le saumon coho (O. kisutch) et la truite arc-en-ciel (Salmo gairdneri) les effets toxiques combines que peuvent avoir les concentrations de zinc, de cuivre et de cadmium dissous et en suspension trouvés dans des échantillons d'eau prélevés dans la rivière Campbell, sur l'île Vancouver (C.-B.) au cours des 12 dernières années (1971-1982). Comme on n'a aucune preuve directe de la toxicité de l'eau de la rivière Campbell, l'évaluation repose sur des données pertinentes tirées des travaux toxicologiques actuels. Il y a eu une augmentation constante des concentrations de zinc dans l'eau de la rivière, qui sont passés d'environ 0,007 mg/L au début de 1971 à environ 0,048 mg/L à la fin de 1980. On n'a pu déceler, au cours de la même période, d'augmentation constante dans les concentrations de cuivre. Cependant, étant donné une dureté moyenne de l'eau de 21,5 (mesuré en mg/L de $CaCO_3$) (1971-1982) les concentrations moyennes de zinc et de cuivre dans la rivière au cours des dernières années ont été respectivement de 0,0464 mg/L (1981-1982) et de 0,0021 mg/L (1980-1982) tandis que celles de cadmium se sont situées, depuis 1971, en dessous de la limite de détection (0,0005 mg/L). Au cours des dernières années (1980-1981), les "pointes" de concentrations de zinc et de cuivre ont atteint respectivement un maximum de 0,066 mg/L et de 0,0035 mg/L. On croit que les concentrations des métaux pourraient avoir quelque peu diminué depuis le milieu de 1981, mais il faudrait effectuer d'autres analyses pour en être certain.

L'évaluation des concentrations CL50-96 h et les données relatives aux concentrations maximales acceptables de substances toxiques (CMAT), qui sont tirées de la documentation, sont fournies pour les espèces examinées lors de tests où la dureté de l'eau était voisine de celle de l'eau de la rivière Campbell. À partir de ces données, on estime les valeurs des CMAT et on les compare aux concentrations des métaux trouvés dans les échantillons d'eau. En admettant que les effets toxiques s'additionnent (Zn + Cu), on en déduit, pour les espèces concernées, le nombre d'unités toxiques (UT) correspondant aux concentrations (Zn + Cu) présentes dans les échantillons. D'après les concentrations moyennes de métaux, on en conclut que l'eau de la rivière Campbell était toxique pour les jeunes S. gairdneri non exposés, c'est-à-dire pour ceux qui proviennent de tributaires ou de la pisciculture Quinsam, pour la plupart de la periode entre fevrier 1980 et septembre 1981. On pense que les jeunes salmonidés déjà exposés, dont l'incubation au stade d'oeuf et l'éclosion ont eu lieu dans la rivière, sont plus résistants aux métaux lourds, probablement grâce à l'action de métallothionéines qui ont tendance à lier ensemble les ions des métaux lourds et à neutraliser leur effet toxique. Il semble que l'eau de la rivière Campbell no soit pas toxique pour les poissons qui ont déjà été exposés mais on ne peut, pour le moment, évaluer le

degré de risque. Au cours des deux dernières années (1980-1981), il semble que les concentrations de zinc et de ciuvre aient atteint un seuil cirtique pour les jeunes salmonidés non exposés au préalable aux substances toxiques. Au cours de cette période, on évalue que les concentrations des métaux lourds présents dans les échantillons d'eau étaient toxiques dans 3 cas sur 38 pour le saumon quinnat, dans 30 cas sur 40 pour la truite arc-en-ciel et dans 0 cas sur 38 pour le saumon coho. De façon générale, on estime que la sensibilité des trois espèces à la toxicité par les métaux lourds peut s'exprimer comme suit: S. gairdneri saumon quinnat saumon coho. Bien que les poissons non exposés (tributaire et piscifacture) et exposés (rivière Campbell) montrent des différences de sensibilité, de petites différences observées dans les caractéristiques de l'eau résultant de changements survenus en amont (par exemple: diminution de la dureté de l'eau ou du pH) pourraient accroître le potentiel toxique des concentrations actuelles de métaux lourds dans l'eau de la rivière. On en conclut que la qualité de l'eau de la rivière Campbell est devenue critique pour le bien-être des salmonidés indigènes. D'après les concentrations actuelles de cuivre et la tendance à la hausse de la concentration de zinc, on estime que la rivière Campbell est devenue, au début de 1980, toxique pour les jeunes S. gairdneri provenant du tributaire et de la piscifacture Quinsam. Étant donné la gravité manifeste de la situation actuelle, nous formulons les recommandations suivantes: (1) des correctifs devraient immédiatement être apportés pour portéger les salmonidés indigènes présents dans la rivière Campbell en réduisant les concentrations actuelles de zinc et de cuivre dans la rivière et (2) des recherches devraient être entreprises pour déterminer les concentrations maximales acceptables de métaux lourds (Zn + Cu + Cd) pour les salmonidés indigènes non exposés et exposés ainsi que pour des organismes représentatifs (algues, invertébrés) de la chaîne alimentaire du bassin de drainage de la rivière Campbell.

1. INTRODUCTION

Within the last two years increasing attention has been directed to the levels of heavy metals--particularly zinc (Zn), copper (Cu) and cadmium (Cd)--found in the Campbell River and its headwaters. This report examines the information available from water samples taken in the lower Campbell River over the past 12 years. As little direct evidence is available on the toxicological status of the Campbell River, an assessment is made of the potential biological effects of these metals based on data from the current literature. A perspective is provided regarding the potential joint toxicity of existing heavy metal loads to three salmonids native to the river--the chinook (Oncorhynchus tshawytscha) and coho (O. kisutch) salmon and the steelhead trout (Salmo gairdneri)[1].

The Campbell River, with a drainage area of about 679 sq miles (1759 sq km), drains into Discovery Passage on the eastern side of north-central Vancouver Island (Fig. 1). It drains a system of lakes (John Hart, McNar, Lower Campbell, Upper Campbell, Buttle), with an average discharge of 3430 cu ft.sec (97 cuM/sec) and maximum and minimum flows of 29500 and 446 CFS (about 835 and 13 cuM/sec). There is a falls impassable to upstream fish migration 3.5 miles (5.6 km) from the river mouth. Above the falls hydroelectric development has occurred with completion of storage dams and powerhouse facilities in 1947. A diversion dam, constructed in 1957, diverts water from the adjacent Salmon River watershed into Brewster Lake, thence into the upper Campbell system.

1.1 Salmonid Fishes in the Campbell River System

The lower 3.5 miles (5.6 km) of the Campbell River support substantial runs of Pacific salmon (Oncorhynchus). Historically the Campbell River tidewater area has had an internationally known sport fishery for chinook salmon for many years. Between 1947 and 1975, average (minimum, maximum) spawning runs in the Campbell River were estimated as: chinook--3600 (750-8000), coho--1220 (25-2500), chum (O. keta)--2850 (750-15000) and pink (O. gorbuscha)--2150 (75-7500). In addition, an average of 980 (75-1500) steelhead trout (Salmo gairdneri) were estimated to spawn in the river annually between 1947 and 1958. Small number of spawning sockeye have been reported (1968-1975), averaging about 115 per year (25-200). The Quinsam River, a major tributary to the lower Campbell River, also supports Pacific salmon spawning runs. From 1957 to 1975, estimates of adult spawners included: chinook--60 (5-200), coho--4180 (750-12000), chum--760 (20-3500) and pink--4200 (750-7500). Steelhead also use the river as well as small numbers of sockeye--65 (25-200). Two other tributary streams also support coho--Nunns Creek and Kingfisher Creek (Marshall et al. 1977).

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[1]Salmo gairdneri comprises two variants, the sea-run steelhead and the non-migratory rainbow. Where no distinction is made between them, they will be called, collectively, S. gairdneri.

A further source of salmon is the Quinsam Hatchery on the Quinsam River; it has been operating since 1974. In 1980, 2071 chinook returned to the Campbell and Quinsam rivers; 58% of these were of hatchery origin. Coho, pink salmon and steelhead are being raised and liberated by the hatchery. In 1980, 39,000 coho returned to the Quinsam River, the largest return of coho since the hatchery began operations in 1974 (Glover 1982).

In addition to the five species of Pacific salmon and steelhead, the lower Campbell River contains populations of dolly varden (Salvelinus malma), resident rainbow trout (Salmo gairdneri) and coastal cutthroat trout (Salmo clarki clarki). These same species are found in the lake system upstream, as well as the non-migrating variant of the sockeye--the kokanee(R. Marken, Campbell River, personal comm.).

1.2 Mining Activities, Buttle Lake

In 1966, Westmin Resources Ltd (Western Mines Ltd) began operation of a mine and concentrator (copper, lead, zinc) located at the southern (upstream) end of Buttle Lake on the headwaters of the Campbell system. Since 1967, tailings from the concentrator have been discharged under permit into the south end of Buttle Lake by submerged outfall. Up to 1974, no major detrimental effects resulting from tailings discharge into the lake could be identified, as reviewed by Clark (1980), although some aspects of mining operations had resulted in changed conditions outside of permit limits. In addition, data from various sampling points in Buttle Lake indicated that turbidity, solids, and copper and zinc levels in Buttle Lake water had increased significantly at lower depths. More recently a trend toward increasing zinc concentration was identified (1978). Clark (1980) concluded that zinc in Buttle Lake water had increased significantly over the last decade (surface and at depth), along with increased concentrations of copper in surface waters. Copper concentrations also showed increasing trends at some locations at 30-M depths. Clark concluded that copper and zinc appear to pose a potential threat to aquatic life in Buttle Lake, and perhaps to portions of upper Campbell Lake. He recommended an urgent evaluation of the environmental impact of the Westmin Resources discharge in detail to determine possible remedial action. Recommended for investigation were Buttle Lake, tributary creeks, Upper Campbell Lake, and possibly downstream waters.

1.3 Analysis of Water Samples, Lower Campbell River

Over the period 1971-1982 water samples have been collected and analyzed by two agencies: The Provincial Ministry of Environment (ME) and the Federal Department of Fisheries and Oceans (DFO) (McLean, 1982a,b). In general, ME sampling covers the period from 1971 to the present; DFO sampling is more intensive, from 1980 until the present. Three types of analysis have occurred. The ME sampling has analyzed for total and dissolved metals; DFO sampling has concentrated on extractable analyses. The differences are as follows.

Analysis of a water sample for total zinc measures all zinc in the sample, whether it is imbedded in the matrix of finely divided solids in the water, adsorbed on the surface of those solids, or dissolved in the water. Dissolved zinc analyses measure only the dissolved metal concentrations, while extractable analyses measure the sum of dissolved and adsorbed zinc. Differences in the amounts detected by each method depend to an extent on other characteristics of the water.

Most important from a biological viewpoint, however, is the fact that heavy metal toxicity shows a higher correlation with dissolved metals than with total metal analyses (dissolved+particulate+adsorbed). The adsorbed fraction, measured by extractable analyses (dissolved+adsorbed), would add a small amount to the estimates of dissolved metal. Although it would be preferable to have all metal estimates as the dissolved fraction, extractable values are accepted as reasonable but slightly higher estimates of the dissolved fraction.

The first set of available data is that taken by the Ministry of Environment (BC) (Table 1) at site 0120820[1], upstream from the confluence of the Quinsam and Campbell rivers (1971-1981). As an estimate of the amounts of metal existing in dissolved form prior to 1980, comparisons were made for paired dissolved and total estimates of Zn, Cu and Cd in the period 13 Oct 77-17 Nov 81 (Table 1). Estimates below detection levels (e.g. <0.01) are omitted (a value of <0.01 could be 0.009 or it could be zero). For the period considered, average Zn, Cu and Cd levels are

Metal	No of pairs	Dissolved (mg/L)	Total (mg/L)	(Diss/Total) X 100 (%)
Zn	12	0.0393	0.0434	90.6
Cu	10	0.0017	0.0024	70.8
Cd	7	<0.0005	<0.0005	-

That is, 90.6% of total zinc in the samples compared is estimated to be in the dissolved form. Similarly, 70.8% of the Cu is estimated to be in the dissolved fraction. Such information can not be obtained for Cd; the amounts present appear to be below detection limits for the method of analysis used. The estimates obtained for Zn and Cu may be used as correction factors to estimate dissolved metal values for the earlier sampling times when total Zn and Cu were obtained, but dissolved Zn and Cu were not. Such derived values for dissolved Zn and Cu are approximations; however they do reduce the quantities involved and therefore are considered to be conservative. A second series of determinations was obtained by the Department of Fisheries and Oceans (Table 2). These extractable values are more intensive for the 1980-81 period.

To obtain a measure of potential changes in Zn and Cu concentrations occurring over time, the data in Tables 1 and 2 are consolidated in Table 3. The derived estimates of dissolved Zn and Cu (Table 3) are those using the correction factors obtained earlier. Table 3 contains the following:

[1]Sample site numbers are those shown by Clark (1980).

(1) original ME and DFO data on water hardness

(2) estimates for Zn (mg/L)
-original ME data for dissolved zinc and (in brackets) estimates of dissolved zinc from analyses for (total zinc X 0.906)
-original DFO data for extractable zinc

(3) estimates for Cu (mg/L)
-original ME data for dissolved copper and (in brackets) estimates for dissolved copper from analyses for (total copper X .708)
-original DFO estimates for extractable copper

Although a number of the estimates of dissolved metals are derived and therefore are indirect estimates, the combined ME and DFO dissolved and extractable estimates for Zn and Cu do provide the opportunity to examine all of the data for trends over the 1971-81 period. There is a suggestion that water hardness (Fig. 2) may have undergone a reduction from 23-24 (as mg/L $CaCO_3$) from 1971 to a minimum around 19-20 about 1976, with a return thereafter to original levels around 1980. Zinc (Fig. 3) shows a consistent increase from 1971 until about the end of 1980. Thereafter there is a suggested downturn during 1981 from approximately .050 mg/L to about .035 mg/L. Copper (Fig. 4) is more difficult to assess. During the 1980-81 period of more intensive sampling there is considerable variability in the copper levels determined. Prior to 1980 the data probably are too sparce to merit further comment.

The next question is how best to present the data on water characteristics. The majority of analyses for water hardness range between 18 and 26 (as mg/L $CaCO_3$). Although the suggestion of a trend exists (a minimum near 1976) there is substantial scatter. The water is soft and it is presumed the mean (21.5 mg/L as $CaCO_3$) is a reasonable representation. The pattern for zinc is more complex. There appears to be a consistent rise in zinc concentration between 1971 and the end of 1980. This trend can be characterized by the polynomial (Fig. 3):

$$Y = 6.73 + 0.004433D + 0.0000019D^2 \quad (n = 39,\ r = 0.804,\ P<0.01)$$

where Y = µg/L zinc
D = days from January 1971.

The trend line is halted at 31 December 1980; in 1981 there appears to be a basic change in the relation of zinc concentration with time. Although there is considerable scatter, there is a suggestion of a reduction in zinc concentration in the latter half of 1981. Whether this apparent downward trend is real or not will be borne out in time only by further sampling. Analyses in 1981 include both the highest and lowest estimates of zinc concentration over the last several years. Therefore the best estimate of zinc concentration in 1981 is taken as the mean, 0.0464 mg/L (dissolved and extractable analyses, Table 3). There are few data points for copper prior to 1980. Those existing (Fig. 4) occur predominately at two levels suggesting that they were subject to limitations of sensitivity of measurement. No trends can be recognized in copper concentration prior to 1980. The more extensive data in 1980-1981 shows considerable scatter, and the mean concentration (0.0021 mg/L dissolved and extractable copper) is taken as a

means of representing copper concentration in that period. The results of water quality analyses for water hardness, Zn, Cu and Cd concentrations in Campbell River are summarized in Table 4.

1.4 Effects of Heavy Metals on Aquatic Organisims

Many metals are essential for living processes. Haemoglobin, essential to the transport of oxygen in the blood of higher organisms, contains iron in its structure. In a number of lower organisms the place of iron in oxygen transport is taken by copper. Some vitamins and a number of enzymes include heavy metals in their structures or reactions. Yet, as is common with many elements and reactive groups, while limited amounts may be required for normal function, too much can be deleterious, toxic, or life-threatening. Other heavy metals normally are not involved in metabolic processes and generally tend to inhibit enzyme reactions. In this report attention is directed mainly to zinc (Zn) and copper (Cu); lead (Pb) and cadmium (Cd) also play a role, particularly Cd.

1.4.1 Toxicity of heavy metals

Judging the deleterious influence of heavy metals on living systems requires some comment on the techniques employed. During the last 40 years there have been substantial developments in toxicology, the technology dealing with the measurement of response of living systems to unnatural levels of natural or foreign materials in an animal's living space. Early work concentrated on the acute or short-term effect of toxicants on animal survival. Prior to 1960 emphasis was placed on determining the relation between toxicant concentration and the exposure time required to elicit a response--usually mortality (Fig. 5). Using such techniques, much effort was expended measuring the relative toxicity of an expanding array of insecticides or pesticides and the unwanted effects these could have on non-target organisms. In the early 1960's attention began to turn to more general consideration of the influence of toxicants on organisms in their normal environments. This ecological approach to pollution of natural environments continues to command major attention.

In the last two decades attention also has expanded beyond the concern for measurement of short-term toxic effects to longer term or chronic effects of pollutants. It became evident that short-term measures of lethality were insufficient in attempts to protect or provide for the ecological requirements of living organisms. Attention expanded to consider dosages and response systems that might measure deleterious effects to the animal without necessarily being lethal. For fishes, whole-organism responses elicited by pollutants, which may have sublethal deleterious effects, include those affecting:

- -behaviour, including avoidance, attraction, orientation
- -temperature selection
- -locomotion, swimming performance
- -chemosensory response
- -cough frequency (clearing response to irritation)
- -feeding, metabolism, growth

-osmoreglation, blood characteristics
-migration
-reproduction, embryonic development

Since the 1940's efforts have been made to develop standard methods for the examination of short-term, acute, lethal responses of living organisms. Short-term tests generally produce a concentration-response relation resembling a rectangular hyperbola (Fig. 5), relating time to death in half of the sample of fish exposed to the test concentration. Comparisons then are possible for concentrations that kill half the test subjects in a standard period of, say, 24, 48, or 96 h. The latter has become a standard measure--the 96-h LC50--the concentration that produces its effect in one-half the exposed subjects within a period of 96 h (4 days). Distributions of response such as that in Fig. 5 would include the 96-h LC50. However, one notes that the LC50 response continues to occur over longer exposures to lower concentrations. Ultimately the upper limit of the curve becomes perpendicular to the concentration axis as the asymptote (limit) is approached. The limiting concentration in short-term tests often is associated with an exposure time of 5-7 days. The limiting concentration has been termed the incipient lethal level (ILL), the concentration that would result in a response in one-half the test subjects after an infinitely long (in short-term tests, 7-day) exposure.

In contrast to short-term tests involving lethal responses, long-term tests of chronic toxicity may examine non-lethal responses or lethal responses over much more extended periods of exposure. Fig. 6 shows a comparison of the sensitivity of nine response systems in terms of threshold concentrations (ILL's), using zinc as the mediator. There, chronic toxicity tests are seen to have lower thresholds than acute (short-term) tests, while aspects of reproductive, growth or locomotor responses are the most sensitive estimators of chronic effects. The latter three exemplify the situation where the organism may not be killed directly, but where its reproductive potential, growth, or ability to avoid predation or capture food may be impaired. Thus, where sublethal effects are involved, reduced probability for survival of the individual reduces survival potential of the population. More recent examinations of chronic toxicity (Fig. 7) show that short-term tests may miss entire episodes of response occurring after prolonged exposure to low concentrations. Fig. 7 is indicative that toxic action may occur in several modes, depending on the sensitivity of the various biological systems affected. In addition, chronic tests are being conducted over increasingly protracted exposure periods, and in some instances over several generations. In such instances it is found that some life stages of an organism are more sensitive to toxicants than others. For example, swim-up fry stages of chinook salmon are most susceptible (alevin, swim-up fry, parr, smolt) (Chapman 1978a). In addition, LC10 concentrations (rather than LC50's) may be estimated--those concentrations of toxicants that produce a response in 10% of a sample of fish after a long (e.g. 200-h, 400-h, 3-month) period of exposure.

1.4.2 Influence of water hardness on toxicity

Water hardness is a measure of divalent metal ions present in water. These generally are calcium (Ca^{2+}) and magnesium (Mg^{2+}), although iron, strontium and manganese also may contribute when appreciable amounts are

present (Environmental Protection Agency 1977). Hardness is an estimate of the amount of such ions present, reported as the equivalent in mg/L of $CaCO_3$; it also is correlated with other water characteristics such as pH, and alkalinity (a water's capacity to neutralize acid or base). When the latter are held constant, the resistance of fishes to heavy metals generally increases as hardness increases. The toxicity of heavy metals is related to the presence of ionic metal species rather than to the total metal present, and the various ionic species that can be present vary with hardness and pH. For example for copper at pH levels below 7, Cu^{2+} is the predominate form; above pH 7, hydroxides and carbonates are most prevalent (Howarth and Spraque 1978). Zinc in waters of low alkalinity, such as the Campbell River, would be predominately in the forms $ZnHCO_3^+$, $ZnCO_3$ and Zn^{2+} (NRCC 1981). Campbell River water is very "soft" (low hardness level) and unit quantities of Zn and Cu in solution would be toxic at relatively low levels, compared with other waters of higher pH or hardness.

1.4.3 Toxicity of mixtures of Zn, Cu and Cd

In the 1960's numerous investigators began to examine the simultaneous action of several toxicants to fishes. It was found that the action of Zn and Cu was additive in Atlantic salmon (Salmo salar) and rainbow trout (Salmo gairdneri). That is, the two heavy metals had a similar action, acting as if they were one metal species. Therefore concentrations of Zn and Cu, taken as fractions of the incipient lethal concentration (ILL), could be summed as "toxic units" (TU) to give the ILL of the Zn-Cu mixture:

$$C_{Zn}/ILL_{Zn} + C_{Cu}/ILL_{Cu} = C_{Zn} + C_{Cu}/ILL_{Cu+Zn} \qquad (1)$$

where C_{Zn} = concentration of zinc

C_{Cu} = concentration of copper

ILL_{Zn+Cu} = incipient lethal concentration of zinc + copper

Dividing both sides of equation (1) by $C_{Zn} + C_{Cu}$ gives the expression for ILL_{Zn+Cu}.

$$1/ILL_{Cu+Zn} = C_{Zn}/(C_{Zn} + C_{Cu})ILL_{Zn} + C_{Cu}/(C_{Zn} + C_{Cu})ILL_{Cu} \qquad (2)$$

This relationship was proven by Sprague (1964) for Atlantic salmon. Similar additivity for Zn and Cu has been demonstrated for chinook salmon eggs, alevins and fry (Finlayson and Verrue 1980). The case for cadmium is much less well understood. Cadmium is toxic at very low concentrations. Yet in combination with zinc and copper the influence of cadmium on toxicity has been found to range from more-than-additive to less-than-additive, depending on the species of fish, the response system tested and the Zn:Cu:Cd ratios involved (EIFAC 1977, 1980). It is known that the toxic effects of cadmium are reduced when zinc is present (Ray and Coffin 1977) and that the affinities of Zn^{2+} and Cd^{2+} ions for sulphide, RS^-, NH_3, imidazole, acetate and glycinate groups, typical of those found in proteins, are quite similar (Hochster and Quastel 1963). Hence Zn^{2+} and Cd^{2+} may compete for binding sites, saturating sulphide

groups first, any excess then associating with other binding sites according to their relative affinities. If this were the situation, then Zn and Cd would bind to protein approximately in proportion to their relative concentrations. Thus, if cadmium-zinc ratios were near 0.0005:0.0464 or about 1:100 (Table 4), then one might expect the response to such a mixture to appear as if it were a response to zinc alone. A conservative method of accounting for Cd toxicity then would be to consider cadmium toxicity as additive when zinc is present at its ILL. In view of the lack of more quantitative information, this is considered to be a reasonable approach in the present situation.

1.4.4 Defining "safe" and "deleterious" concentrations

Ecologically, short-term (eg 96-h LC50) toxicity tests are not in themselves measures of concentrations that would be harmless after long exposure to a toxicant. For a given toxicant, however, the ratio can be established between the highest concentration that is just without deleterious effect (the maximum acceptable toxicant concentration or MATC) and that obtained for a standard 96-h LC50 test. This ratio,

$$\frac{\text{MATC}}{\text{96-h LC50}} = \text{Application factor (AF)},$$

provides a ratio that can be used to estimate a concentration of the toxicant estimated to be safe in terms of the test organism and similar environmental circumstances (eg 96-h LC50 X AF = safe concentration).

It is assumed from section 1.4.3 that Zn and Cu effects are additive, and that Cd effects are additive when zinc is present at its incipient lethal level. Under these conditions it is assumed that toxic effects can be represented as follows:

$$Ca/La + Cb/Lb + \ldots Cn/Ln \geq 1.0, \qquad (3)$$

where Ca, Cb, ..., Cn are the measured or expected concentrations of toxicants a, b, ..., n in the water, and La, Lb, ..., Ln are the recommended or "safe" concentrations--those derived for each a, b, ..., n toxicant as the (96-h LC50 X AF) concentrations. For each L component may be substituted the MATC (maximum acceptable toxicant concentration), if these are known. Further, NAS/NAE (1974) recommends that any fraction (C_k/L_k) with a value <0.2 should be removed from the calculation, to "remove natural trace concentrations and low nonadditive concentrations from the formula". In the present instance fractions <0.2 have been retained for Zn and Cu. The argument followed is that the heavy metal loads being examined are closer to toxic levels than to background levels: under these circumstances small additions are considered more likely to have biological consequences. When the sum of the fractions in equation (1) is <1, the mixture should be safe; conversely if the sum is ≥1, the mixture should be considered toxic.

The following is an example of how toxicity would be predicted using equation (3). The data in the example, not included in Tables 1-3, are for a sample of Campbell River water taken February 4, 1981 by the Fish and Wildlife

Branch (ME) and analyzed by the Waste Management Branch (ME). These data are used here as they provide a full illustration of the use of toxic units, the level of Cd being above the limits for detection. Accordingly,

Zn (total) 0.060 mg/L
Cu (total) 0.018 mg/L
Cd (total) 0.0009 mg/L

Levels of dissolved metals estimated:

Zn 0.906 x .060 = 0.054 mg/L
Cu 0.708 x .018 = 0.0127 mg/L
Cd assumed 0.0009 mg/L

Additive toxic units (equation (3)):

$$TU = Ca/La + Cb/Lb + Cc/Lc$$

For S. gairdneri:

Zn: C_a = 0.054 mg/L L_a = 0.045 mg/L (Table 15)
Cu: C_b = 0.0127 mg/L L_b = 0.011 mg/L (Table 15)
Cd: C_c = 0.0009 mg/L L_c = 0.0007 mg/L (Table 15)

For Zn + Cu, since

$$C_a/L_a + C_b/L_b = .054/.045 + .0127/.011 = 2.35 > 1$$

Then, adding Cd:

$$TU = 2.35 + 0.0009/0.0007 = 3.64$$

That is, additive toxicity (TU) is 3.6 times the threshold for toxic effects in naive juveniles of S. gairdneri, (see Sections 2.8-2.10).

To use such a prediction system requires the examination of 96-h LC50 estimates, those for MATC's, the computation of application factors for species of fishes native to the waters in question and the selection of such data where background water quality conditions are similar to those for the Campbell River. The available data, summarized from other sources, is presented in the following section.

2. SUMMARY OF THE LITERATURE ON THE TOXICITY OF Zn, Cu AND Cd

Much of the available literature has been well summarized previously. Such sources include McKee and Wolf (1963), National Academy of Sciences, National Academy of Engineering (1974), Environmental Protection Agency (1977), Ray and Coffin (1979), Thurston et al. (1979), European Inland Fisheries Advisory Commission (1973, 1977, 1980), NRCC (1979, 1981). Most of the references shown in the following tables are from those review sources.

Useful specific information also was obtained from a number of original papers. These include Sprague and Ramsay (1965), McKim and Benoit (1971), Sinley et al. (1974), Nehring and Goettl (1974), Lorz et al. (1978), Chapman (1978a,b), Eaton et al. (1978), Chapman and Stevens (1978), Howarth and Sprague (1978) and Finlayson and Verrue (1980). The data reported are confined largely to those involving salmonid species found in the Campbell River. In some instances data on other salmonids are included, where those data provide information on the arguments basic to considerations of joint toxicity of heavy metals.

2.1 Short-term toxicity of zinc to salmonids

Table 5 summarized the results of acute toxicity tests for coho, sockeye and chinook salmon and for Salmo gairdneri (steelhead-rainbow trout). From the table it can be seen, in general, that toxicity of zinc declines with increasing water hardness, and that toxicity varies in relation to size or age of the test organism.

2.2 Long-term toxicity of zinc to salmonids

Table 6 lists lethal and sublethal effects of exposure to zinc found in chronic tests. Again, and in the tables to follow, toxicity declines with increase in water hardness. Other than the work of Sinley et al. (1974), little evidence is available from chronic tests that would provide data on threshold concentrations of zinc applicable in the current situation.

2.3 Toxicity of zinc to representative fish food organisms

Table 7 lists results of short-term tests of the toxicity of zinc to several organisms representative of food items for young salmonids in the natural environment. Toxicity of zinc to the organisms shown appears to be of the same order of magnitude (1-day LC50's) as for juvenile salmonids (4-day LC50's).

2.4 Short-term toxicity of copper to salmonids

Table 8 summarizes the results of a number of short-term tests of copper to salmonids. As for zinc, copper toxicity declines with increasing water hardness, and LC50's vary with size or age of the test organism. The paper by Howarth and Sprague (1978) has been used to estimate a 96-h LC50 for copper for S. gairdneri in terms of the average pH and water hardness of the Campbell River system.

2.5 Long-term toxicity of copper to salmonids

Table 9 presents lethal and sublethal effects of exposure to copper as found in chronic tests. Note that the threshold for avoidance of copper in Atlantic salmon occurs at about 0.13 of the 96-h LC50 (Table 8) (Sprague and Ramsay 1965).

2.6 Toxicity of copper to representative fish food organisms

Table 10 lists the results of tests of the toxicity of copper to several fish food organisms and other organisms lower in the food web. Such organisms appear to be as sensitive or more sensitive to copper, in comparison with the result avaliable for salmonid species (Tables 8,9).

2.7 Toxicity of cadmium to salmonids and fish food organisms

Table 11 summarizes some responses of salmonids and fish food organisms to cadmium. Fishes appear to respond to cadmium differently than to zinc or copper. Over the period of a short-term test to cadmium, mortality may be slight or absent, resulting in an LC50 estimate (mg/L) that may be high numerically. However, in longer-term bioassays mortality occurs and continues to very low concentrations. Indeed, there is some question in some of the literature as to whether the LC50 estimates that have been obtained are a function of the sensitivity of the organism tested or of the method used to measure cadmium concentration. Increased sensitivity of 1-2 orders of magnitude in measuring cadmium would appear to be worthwhile in further studies of cadmium toxicity--based on a current detection limit of 0.0005 mg/L.

2.8 Toxicity of Zn-Cu-Cd mixtures to salmonids

Table 12 summarizes data pertaining to the toxicity of mixtures of heavy metals, primarily to salmonids. The work of Sprague and Ramsay (1965) for Atlantic salmon is indicative of the additive nature of Zn-Cu toxicity. Note that levels of Cu interfering with migration of Atlantic salmon (Sprague 1964) (0.052 of 7-d LC50 of 0.032 mg/L) are very similar for coho in harder water (Lorz and McPherson 1977) (0.07 of 4-d LC50 of 0.071/mg/L). Cadmium produces a les-than-additive response, in combination with Zn and Cu, for the fathead minnow (Eaton 1973).

2.9 Toxicity of Zn, Cu and Cd at different stages of development in salmonids

Table 13 consolidates information on four stages of development regarding toxicity of Zn, Cu and Cd to three species of interest--chinook and coho salmon and S. gairdneri. In general, the swim-up fry appears to be the most sensitive stage with the possible exception of the response of S. gairdneri to Cu. For that species the swim-up fry, parr and smolt are all about equally susceptible to Cu (200-h LC10s).

2.10 Acclimation of fish to heavy metals

A number of investigators (Sinley et al. 1974; Chapman 1978b; Lorz et al. 1978: Finlayson and Ashuckian 1979; Finlayson and Verrue 1980) have noted that previous exposure to sublethal levels of heavy metal increases the resistance to or decreases mortality resulting from exposure to otherwise lethal concentrations. For example, pre-exposure to zinc can result in a

2-fold to 3-fold increase in the 7-day LC50 (EIFAC 1973).[1] Similar results have been noted for cadmium (EIFAC 1977). As a result, naive fish--those not previously exposed to heavy metals--may be less resistant than pre-exposed fish when subjected to a sudden increase in heavy metal concentration. It appears that exposure of fish to heavy metal induces the formation of metallothionein-like proteins, which bind and store or assist in the elimination of the metal (McCarter and Roch, MS; Roch et al. MS).

Hence, naive and pre-exposed fish could have different response thresholds to Zn, Cu, and Cd. In the Campbell River, this could lead to more resistant fish resulting from eggs laid and incubated in the main river. However, those juveniles entering the main river from feeder streams would be classed as naive, including those from the Quinsam River and Quinsam Hatchery. Feeder stream migrants moving into the main river could be expected to respond to heavy metals according to the MATC estimates generated herein.

3. INTERPRETATION

3.1 Estimating "safe" levels of heavy metal concentrations

In section 1.4.4 the concept of Application Factors was defined, a concept leading to estimates of safe levels of metal concentration (eg 96-h LC50 X AF = "safe" level). The Environmental Protection Agency (EPA 1977) recommended application factors as follows:

Zn: 96-h LC50 x 0.01, determined through bioassay using a sensitive resident species

Cu: 96-h LC50 X 0.1, determined through (nonaerated) bioassay using a sensitive resident species

Cd: no AF recommended; instead a maximum of 0.0004 mg/L is thought to be sufficient to protect salmonids in soft waters.

In a critique of the EPA (1977) recommendations, Thurston et al. (1979) considered the AF for zinc (0.01) to be too strigent, and suggested an AF of 0.05 as an alternative for very soft waters (0-75 mg/L $CaCO_3$). Thurston et al. (1979) also argue against the use of the 0.1 application factor for Cu, or any application factor for Cu, although their arguements are difficult to support. The same authors concur that 0.0004 mg/L Cd probably is a safe level of Cd for salmonids. They point out, however, that lower concentrations (0.00017 mg/L in soft water) have caused reproductive impairment in Daphnia magna, a typical fish food organism (Biesinger and Christensen 1972). Hence the 0.0004 mg/L "safe" level for cadmium may err by being too high.

For the above reasons the decision was made to review the available literature for heavy metals, with emphasis on estimates of toxicity for

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[1]Unpublished data of G. A. Chapman, Corvallis Ore. (personal communication), indicates the increase can be 2-fold to 5-fold.

species of salmonids native to the Campbell River. Table 14 consolidates, from the previous tables, those data judged to be most pertinent to the estimation of application factors. Judgement was involved in the selection; however, the absolute values involved appear reasonable. Sufficient data were found to allow consideration of three species: chinook and coho salmon, and S. gairdneri. Selection also was confined to tests involving very soft waters, with a hardness near that of Campbell River water (21.5 mg/L $CaCO_3$). In some instances, the nearest approximation to appropriate data is used (eg response of coho to Cu) where there are few data to choose from. Few "maximum acceptable toxic concentrations" (MATC) are available for the three species; accordingly, in a number of instances the lowest levels of metal concentration at which a minimum response was noted above control levels was accepted as a close estimate of MATC's.

The ratios of MATC (or close equivalent) to 96-h LC50 (Table 14) provide estimates of Application Factors that can be applied to the three species native to the Campbell River. These estimates (Table 15) are conservative when compared with the Application Factors or limiting concentrations recommended by the EPA (1977): Zn--0.01, Cu--0.1, Cd > 0.0004 mg/L. Of course, the MATC or close equivalent (Table 15) is equal to the 96-h LC50 X AF; hence, it becomes the denominator in equation (3) (Section 1.4.4) in estimating the potential toxicity of other samples of heavy metals.

3.2 Predicted effects of Zn, Cu and Cd in the Campbell River

Applying the previous arguments, one may assess the potential toxicity of Zn, Cu and Cd in the Campbell River. We assume the effects of Zn and Cu are additive (Section 1.4.4) and that the effect of Cd is additive if Zn and Cu are jointly toxic (>threshold level) (Section 1.4.3).

3.2.1 Predicted effects of maximum, minimum and average heavy metal loadings

Use of the data from Table 4 is shown in the following example based on mean concentrations of Zn, Cu and Cd:

	(Zn)	(Cu)
(1) chinook:	$\frac{0.0464}{0.077} + \frac{0.0021}{0.0125} = 0.603 + 0.168 = 0.79$	
(2) S. gairdneri	$\frac{0.0464}{0.045} + \frac{0.0021}{0.011} = 1.031 + 0.191 = 1.22$	
(3) coho	$\frac{0.0464}{0.800} + \frac{0.0021}{0.005} = 0.058 + 0.420 = 0.48$	

The influence of Cd would be added to that for Zn + Cu for S. gairdneri. However as the average value of Cd (Table 4) is below detection limits (<0.0005), it cannot be used. All of the estimates from the data in Table 4

are given in Table 16. From Table 16, sensitivity of the three species to Zn + Cu is: S. gairdneri >chinook >coho. Mean (1971-Jan 1982) Zn+Cu levels in the Campbell River are predicted to be toxic to naive S. gairdneri, while maximum levels of Zn + Cu are predicted to be toxic to both S. gairdneri and chinook salmon.

3.2.2 Predicted effects of heavy metals in the Campbell River in the 1980-1981 period

It is recognized that fish do not face average concentrations of heavy metals all of the time. In a range of concentrations, half may be above and half below the average. Of greater concern are those above the average. "Spikes" of higher concentration can occur intermittently, and possibly reach or surpass safe levels of exposure (concentration X time of exposure) even though mean concentrations are non-lethal. Accordingly, the combined data in Table 3 are examined for the period 1980-1981, when sampling intensity on the Campblell River was increased over previous years.

Based on the estimated maximum acceptable toxicant concentrations (Section 3.2.1) the relative susceptibility of the three species of salmonid is:

-for Zn: S. gairdneri >chinook >coho
-for Cu: coho >S. gairdneri >chinook

The additive effects of Zn+Cu for naive fish of the three species, based on the relations developed in Section 3.2.1, are illustrated in Fig. 8. In the figure, 30 of 40 water samples in the 1980-Jan 1982 period are predicted to be toxic to S. gairdneri. In the same period, 3 of 38 samples are predicted to be toxic to chinook salmon, while none of the samples was above the toxic threshold for coho. In the 2-year interval, the most critical period appears to have been that between January and June 1981 when the highest levels of predicted Zn+Cu toxicity occurred. It is of particular interest to note that a major change in predicted toxicity of Campbell River water appears to have begun in late July, 1981. A consistent reduction in predicted toxicity seems to have ocurred between July 1981 and January 1982. Further sampling is required to learn whether or not this is a true and continuing trend.

From the available data it appears that the Campbell River has reached a critical state regarding heavy metal loads and their potential deleterious effect on resident salmonid fishes. It appears that naive juvenile salmonids entering the Campbell River from feeder streams and the Quinsam Hatchery may be at considerable risk. The question remains whether such naive fish could be induced by their entry into metal-containing water to produce metal-binding metallothionein rapidly enough for the fish to acclimate to those levels of exposure prior to the onset of mortality. McCarter and Roch (MS) found that coho juveniles (3-8 g) increased liver metallothionein in direct relation to the logarithum of copper concentration to which they were exposed (11-12°C, water hardness 280 mg/L as $CaCO_3$, alkalinity 270 mg/L, pH 7.6). Further, induced metallothionein reached maximum levels after 4 weeks of exposure to Cu, while 168-h LC50's increased to maximum values after 1-2 weeks exposure and stabilized at lower but increased levels after 4-8 weeks. Such results are instructive and would be of major interest if they could be repeated for soft water (eg 21.5 mg/L $CaCO_3$).

The induction of metallothionein by pre-exposure to heavy metal is assumed to explain the "acclimation" or greater resistance of non-naive fish to metals in solution. Hence fish resulting from eggs spawned, hatched and rearing in the main stem of the Campbell River would be expected to be more resistant to heavy metals that naive (Fig. 8) fish. It is difficult to say whether or not pre-exposed fish would be totally protected from rather than more resistant to metal levels deleterious to naive fish. Evidence from Sinley et al. (1974) for S. gairdneri indicates the threshold for toxic effects (MATC) may rise 3-to 4-fold (0.26 vs 0.07/mg/L Zn for naive fish) and that this increase also will be reflected in the 96-h LC50 (0.430 vs 0.093 mg/L Zn for naive fish). We presume pre-exposed fish would be protected from heavy metals in the main Campbell River by metallothionein-induced acclimation. However, the range of acclimation ability in soft water is not known; until more is known about acclimation to heavy metals in soft water we presume pre-exposed resident fish could be at some risk, particularly if metal levels were to increase beyond those present in 1980-81, or if the pH or hardness of Campbell River water were to show some decline in future. To this should be added a further cautionary note. Analytical detection methods for cadmium (eg 0.0005 mg/L) appear inadequate in view of the fact that levels of 0.0007-0.0015 mg/L are deleterious to various juvenile stages of chinook (Table 13), and in other fishes at 0.001 mg/L (Sangalang and Freeman 1974). Detection limits at least one order of magnitude lower presumably would allow quantification of Cd effects. Interactive effects of Cd with Cu on migration effectiveness and mortality in sea water following Cd-Cu exposure (Lorz et al. 1978) indicate that the possible influence of Cd cannot be disregarded. Potential effects of Cd might have been added to those for Zn and Cu in Fig. 8 if more sensitive data on dissolved Cd in Campbell River water were available.

3.2.3 Trends in predicted toxicity 1971-81

Finally, there is the question of how heavy-metal loads in the Campbell River have varied over the last decade. The tables of water analyses listed earlier (Tables 1-3) indicated a significant rise in Zn levels at least from 1971 to Dec 1980 (Fig. 3). The limited data for Cu are sufficient to provide an estimate of mean dissolved Cu in the Campbell River over the 1980-81 period (0.0021 mg/L), there being no clear trend in the earlier period. In section 3.1 a trend line was established for Zn; beyond 1980 there appears to be a major change in the trend--at least beyond mid-1981. Assuming an average Cu concentration of 0.0021 mg/L in the Campbell River, and accepting the trend for Zn (Fig. 3), the time may be estimated at which the additive Zn+Cu load would reach a toxic threshold if:

$$(X_{Zn}/L_{Zn}) + (C_{Cu}/L_{Cu}) = 1$$

then for (Zn) (Cu)

(1) chinook: $\frac{X}{0.077} + \frac{0.0021}{0.0125} = 1$, X = 0.064 mg/L Zn

(2) S. gairdneri $\frac{X}{0.045} + \frac{0.0021}{0.011} = 1$, X = 0.036 mg/L Zn

(3) coho $\frac{X}{0.800} + \frac{0.0021}{0.005} = 1$, X = 0.464 mg/L Zn

Inserting these values in the trend equation (Section 1.3) provides an estimate of when an additive Zn+Cu toxic threshold would be reached in the Campbell River (μg/L Zn). Solving for D in the polynomial

$$0.0000019D^2 + 0.004433D + (6.73 - X) = 0$$

(1) chinook: D = 4446 days from 1 Jan 71 = 12.2 years
(2) S. gairdneri: D = 2928 days from 1 Jan 71 = 8.0 years
(3) coho: D = 14390 days from 1 Jan 71 = 39.4 years

The calculations suggest that if Zn concentrations were to continue to increase according to their 10-year trend, and if Cu retains its current average concentration, then toxic thresholds (Zn + Cu) for naive fish would be reached:

(1) chinook: early 1983
(2) S. gairdneri: February 1980
(3) coho: (2010)

As indicated previously, there is a suggested change in the trend line for Zn in the Campbell River beyond approximately 1980. Whether or not the apparent pause or reversal in upward-trending Zn concentrations is real will become clearer with further sampling of Campbell River water. Because of the apparent change in the trend line for Zn concentration and the fact that estimates of toxicity of Campbell River water to chinook and coho juveniles involve extrapolations beyond the current chemical data, future predictions for these two species are considered unwarranted until more data become available. However, the date at which the Campbell River is predicted to have become toxic to juveniles of S. gairdneri (February 1980) precedes the apparent end of the upward trend in Zn concentration. In terms of predicted toxicity, the conclusion must be drawn that the Campbell River became toxic to naive S. gairdneri juveniles early in 1980 and remained in that condition for a substantial portion of the following 2-yr period. In addition, Campbell River water is predicted to have been toxic to naive chinook juveniles on occasion (Fig. 8) in the same 2-yr period. These findings attest to the potentially serious nature of the problem and the need for careful attention to future control and assessment of heavy metal loads in the Campbell River.

4.1 CONCLUSIONS

(1) The potential effects of heavy metals, particularly zinc, copper and cadmium, have been reviewed in relation to three salmonids native to the Campbell River system--the chinook and coho salmon and the steelhead (rainbow) trout Salmo gairdneri.

(2) Virtually no direct evidence is available regarding potential deleterious effects of Campbell River water on native salmonid species. Nevertheless, a

substantial amount of information pertinent to the Campbell River situation is available from other sources. Concepts involving the joint action of Zn, Cu and Cd, and a number of specific investigations involving the responses of the three species in question to those metals, have been assembled to provide a means of indirect prediction of the presence, absence or levels of deleterious effects of Zn, Cu and Cd in the Campbell River from 1971 until the present (January 1982). In all instances where judgement has been involved, a conservative approach has been adopted to avoid the possibility of overstating the problem.

(3) For the period from 1971 until 1981 there has been a consistent and significant increase in levels of dissolved zinc in the Campbell River. The limited data on dissolved Cu concentrations in the Campbell River prior to 1980 provide little evidence for the estimation of levels or trends in that earlier period. Levels of dissolved cadmium in Campbell River water have been below analytical detection limits since 1971. Unfortunately those detection limits (0.0005 mg/L) are near the threshold for toxic effect. As a result litle can be said quantitatively about the potential influence of cadmium on the three salmonid species in question.

(4) In the 1980-1981 period more extensive sampling of Campbell River water occurred. There is a strong suggestion that maximum zinc levels (1971-1981) occurred in the first half of 1981 and that zinc levels may have begun to decline thereafter. Whether or not this latter apparent decline is true and continuous will only be evident through continued sampling in the future. Copper levels in the Campbell River during 1980-1981 were highly variable but appear on the average to be higher than those in previous years. The average of dissolved and extractable levels during 1980-1981 (mg/L) were:

Zn--0.0464, Cu--0.0021.

(5) Estimates of 96-h LC50 concentrations and maximum acceptable toxicant concentrations (MATC) were marshalled from the literature from investigations in which water hardness was near that of the Campbell River (average 21.5 mg/L as $CaCO_3$ for the period 1971-Jan 1981). From these, Application Factors were derived. These were found to be conservative, as follows:

	Application Factor			
Metal	Chinook	Derived *S. gairdneri*	Coho	Recommended (EPA 1976)
Zn	0.79	0.49	0.71	0.01
Cu	0.66	0.36	0.09	0.1
Cd	0.67	0.64	0.38	(0.0004 mg/L)

From the derived factors and available 96-h LC50 estimates, final estimates were obtained of maximum acceptable toxicant concentrations. These were (mg/L):

Metal	Chinook	S. gairdneri	Coho
Zn	0.077	0.045	0.800[1]
Cu	0.0125	0.011	0.005
Cd	0.0012	0.0007	0.0039

[1]See footnote, Table 15.

(6) Assuming additivity of toxic effect (Zn+Cu) on the basis of results of other investigations, relative susceptibility to Zn+Cu was found to be

S. gairdneri > chinook > coho.

(7) For the period Jan 1980-Jan 1982 the mean concentrations of Zn and Cu, converted to Toxic Units (TU) were (1) chinook--0.79, (2) S. gairdneri--1.22, (3) coho--0.48. That is, mean Zn+Cu levels in the Campbell River (1980-Jan 1982) are predicted to have been toxic (TU$\geq$1) to S. gairdneri but not to chinook or coho.

(8) In terms of additive (Zn+Cu) effects in the Campbell River (1980-Jan 1982), water samples are predicted to have been toxic (TU$\geq$1.0) as follows:

Species	No. of samples predicted to have been toxic
Chinook	3 of 38
S. gairdneri	30 of 40
Coho	0 of 38

(9) In terms of the 10-year trend of increasing zinc concentration (1971-1981) in the Campbell River and the average copper concentration in the 1980-Jan 1982 period, the Campbell River is predicted to have become toxic to naive juveniles of S. gairdneri early (February) in 1980.

(10) The arguments and conclusions presented regarding the toxicity of heavy metals to salmonids are based on tests in which the fishes involved were naive--that is, not subject to pre-exposure to the metals prior to tests for toxic effects. Pre-exposed fish have the ability to "acclimate" to heavy metals by binding them to metallothionein proteins in the living organism. Naive fish are 2-5 times more susceptible to heavy metal poisoning than pre-exposed fish although this comparison has yet to be substantiated in soft waters such as the Campbell River. Accordingly, eggs spawned in and alevins emerging from Campbell River gravel should be considered "pre-exposed" and somewhat less susceptible to heavy metal toxicity than the previous results would indicate. For example, a doubling of the MATC for the most susceptible species (S. gairdneri) for the potentially most toxic sample of Campbell River water (3 March 1981) shows a TU=0.85, a value high enough to be of concern although, by definition, not toxic. However, pre-exposed fish should be considered at risk until the level of risk can be defined. On the other hand

juvenile salmonids entering the Campbell River from feeder streams, the Quinsam River and the Quinsam Hatchery, would be classed as naive fish. These fish could be at substantial to lethal risk in the Campbell River, particularly juvenile S. gairdneri.

(11) On the basis of the foregoing arguments it appears that heavy metal loads in the Campbell River became critical around the beginning of 1980 for S. gairdneri and have been critical at intervals for chinook. Problems to be anticipated, with naive (not pre-exposed to heavy metals) juveniles in particular, include: direct lethality, disruption of downstream migration of juveniles, delayed mortality of juveniles upon entering salt water, and delayed upstream migration of returning adults. More subtle deleterious effects could include (in juveniles): loss of appetite, reduced growth and various internal metabolic changes. Aquatic invertebrates also are likely to be affected by the heavy metal loads, including organisms upon which juvenile salmonids feed. The situation in the Campbell River appears to be severe enough to warrant immediate remedial action.

4.2 Recommendations

4.2.1 Water sampling and analysis

In terms of the sensitivity of the organisms involved, it is suggested that limits of detection be maintained or adjusted, if possible, so that the following may be obtained:

Metal	Detection limit (µg/L)	Detectable interval (µg/L)
Zn	4	2
Cu	1	0.5
Cd	0.1	0.05

It is recommended that the various agencies involved in sampling in the past meet to decide on a standard sampling procedure and method. Toxicology data suggest analyses for dissolved metal would be most appropriate.

4.2.2 Biological assays

The current results and recommendations are based on pertinent data marshalled from the biological literature. Although they are considered valid, they remain indirect and inferential. An assay program should be developed and implemented, remote or on site, to test the validity of the predictions herein.

4.2.3 Remedial measures

The apparent critical state of heavy metal loads in the Campbell River and the threat they pose to native salmonids in particular, and the Campbell River ecosystem in general, suggest that immediate remedial measures be taken to reduce the heavy metal loads in the Campbell River. It must be

remembered that small changes in other variables under current conditions (such as reduction in hardness or pH) could increase the toxicity of existing heavy metal loads.

The current levels of heavy metals in the lower Campbell system suggest that conditions in the upstream source lakes of the system could be even more critical. Examination of analyses for Buttle Lake (Clark 1980) suggests that indeed this may be the case.

4.2.4 Research needs

To assist in a better understanding of the influence of pre-exposure to heavy metals on resistance to heavy metal toxicity, it is suggested that investigations of metallothionein binding of heavy metals be fostered with the object in mind of estimating changes in 96-h LC50 and MATC concentrations for representative native salmonids stabilized at various sublethal levels of Zn, Cu and Cd in soft waters (hardness of approximately 18-28).

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Table 1. Water hardness (as mg/L $CaCO_3$) and analyses for total and dissolved Zn, Cu and Cd from samples collected by the Provincial Ministry of Environment at site 0120820 (1971-1981) (From computer printout supplied by the Ministy of Environment).

Date	Hardness (mg/L as $CaCO_3$)	Zinc (mg/L)		Copper (mg/L)		Cadmium (mg/L)	
		dissolved	total	dissolved	total	dissolved	total
20 May 71	23.6	0.01	-	<0.01	-	-	-
21 July 71	22.	<0.005	-	<0.01	-	-	-
20 Oct 71	24.	-	0.005	-	<0.01	-	-
11 May 72	19.	-	0.17[c]	-	0.008	-	-
11 July 72	21.	-	0.007	-	<0.001	-	-
7 Feb 73	18.	-	0.01	-	0.001	-	-
25 Apr 73	18.9	-	<0.005	-	<0.001	-	-
7 Aug 73	19.8	-	0.008	-	<0.001	-	-
16 Oct 73	21.3	-	0.008	-	<0.001	-	-
7 Mar 74	18.5	-	<0.005	-	0.012	-	-
29 Apr 74	-	-	0.024	-	0.004	-	-
5 June 74	20.4	-	<0.005	-	0.002	-	-
17 Apr 75	19.2	-	0.014	-	0.002	-	-
7 Oct 75	20.6	-	0.02	-	<0.001	-	-
8 Apr 76	18.5	-	0.023	-	0.001	-	-
30 Sept 76	-	-	0.029	-	0.001	-	-
21 Apr 77	18.5	-	0.03	-	0.002	-	-
13 Oct 77	21.4	0.023	0.029	<0.001	0.001	-	-
19 Apr 78	19.9	-	0.03	-	0.001	-	-
25 Sept 78	23.1	-	0.029	-	0.001	-	-
26 Apr 79	-	-	0.04	-	0.002	-	0.0009
30 Oct 79	21.8	-	0.032	-	<0.001	-	<0.0005
28 Feb 80	-	0.05	0.05	0.002	0.003	-	<0.0005
10 Apr 80	21.5	0.05	0.05	0.001	<0.01	<0.0005	<0.01
18 June 80	21.	0.05[a]	0.05[a]	0.0025	[b] 0.0025[b]	<0.0005	<0.0005
17 Sept 80	-	0.03	0.05	0.001	0.001	-	<0.0005
6 Oct 80	22.8	-	0.05	-	0.001	-	<0.0005
10 Dec 80	-	0.032	0.032	0.002	0.003	<0.0005	<0.0005
19 Feb 81	-	0.05	0.05	0.001	0.002	<0.0005	<0.0005
14 Apr 81	22.9	0.04	0.04	0.002	0.009[c]	<0.0005	<0.0005
24 June 81	-	0.026	0.03	0.001	0.001	<0.0005	<0.0005
20 Aug 81	-	0.04	0.05	0.001	0.004	-	<0.0005
1 Oct 81	21.9	-	0.033	-	0.003	-	<0.0005
17 Nov 81	-	0.03	0.04	0.002	0.003	<0.0005	<0.0005

[a]Average of 2 samples at 0.05 mg/L.
[b]Average of 0.004 and 0.001 mg/L.
[c]Outlier: contamination suspected; not used.

Table 2. Water hardness (as mg/L $CaCO_3$) and determinations of extractable metals (dissolved + adsorbed) in samples collected by the Department of Fisheries and Oceans upstream from the confluence of the Quinsam and Campbell rivers (1974, 1980-82) (McLean 1982a, b)

Date	Hardness (as mg/L $CaCO_3$)	Zn (mg/L) (extractable)	Cu(mg/L) (extractable)	Cd (mg/L) (extractable)
8 Aug 74	14	0.020	0.010[a]	-
20 Aug 74	16	0.040[a]	<0.020	-
23 Aug 74	23.5	0.100[a]	<0.020	-
3 Sept 74	16	-	<0.010	-
6 Sept 74	16	-	-	-
15 Apr 80	24.6	0.047	<0.025	<0.001
22 Apr 80	23.2	0.046	0.0019	<0.001
28 May 80	22.8	0.042	0.0025	<0.001
17 June 80	23.3	0.048	0.0029	<0.0005
24 June 80	20.6	0.049	0.0013	<0.0005
8 July 80	21.6	0.046	<0.010	<0.0005
28 July 80	23.6	0.052	0.0019	<0.0005
25 Aug 80	22.6	0.052	0.0026	<0.0005
9 Sept 80	24.0	0.053	0.0022	<0.0005
15 Oct 80	26.5	0.046	0.0010	<0.0005
17 Nov 80	24.5	0.053	0.0014	<0.0005
27 Nov 80	23.7	0.047	0.0013	<0.0005
11 Dec 80	24.9	0.045	0.0015	<0.0005
22 Dec 80	23.1	0.051	0.0025	<0.0005
12 Jan 81	24.9	0.066	0.0025	<0.0005
2 Feb 81	20.5	0.054	0.0018	<0.0005
19 Feb 81	23.4	0.053	0.0028	<0.0005
3 Mar 81	19.5	0.064	0.0030	<0.0005
24 Mar 81	20.0	0.0498	0.0031	<0.0005
28 Apr 81	21.2	0.0489	0.0029	<0.0005
19 May 81	20.6	0.0616	0.0032	<0.0005
8 June 81	19.3	0.0477	0.0022	<0.0005
29 June 81	20.2	0.0463	0.0035	<0.0005
17 Aug 81	21.2	0.0486	-	<0.0005
1 Sept 81	21.9	0.0488	0.0014	<0.0005
13 Oct 81	22.4	0.0440	0.0014	<0.0005
28 Oct 81	21.5	0.0384	<0.0010	<0.0005
10 Nov 81	23.0	0.0345	0.0022	<0.0005
1 Dec 81	20.4	0.0378	-	<0.0005
13 Jan 82	19.7	0.0348	0.0017	<0.0005

[a]Outlier: contamination suspected; not used.

Table 3. Consolidated list of ME and DFO analyses for zinc and copper in the Campbell River, 20 May 71-13 Jan 82. The ME data in brackets represent dissolved levels computed from total values (see text). The DFO values are analyses for extractable metals (see text). Cadmium is omitted: in almost all cases, analyses gave results below levels of detection.

Date	Days from 1 Jan 1971	Hardness (mg/L)		Zn (mg/L)		Cu (mg/L)	
		ME	DFO	ME	DFO	ME	DFO
20 May 71	140	23.6	-	0.01	-	-	-
21 July 71	202	22.	-	<0.005	-	-	-
20 Oct 71	293	24.	-	(0.0045)	-	-	-
11 May 72	497	19.	-	-	-	(.0057[a])	-
11 July 72	558	21	-	(0.0063)	-	-	-
7 Feb 73	769	18	-	(0.0091)	-	(.0007)	-
25 Apr 73	846	18.9	-	(<0.005)	-	-	-
7 Aug 73	950	19.8	-	(0.0072)	-	-	-
16 Oct 73	1020	21.3	-	(0.0072)	-	-	-
7 Mar 74	1162	18.5	-	(<0.005)	-	(.0085[a])	-
29 Apr 74	1215	-	-	(0.0217)	-	(.0028)	-
5 Jun 74	1252	20.4	-	(<0.005)	-	(.0014)	-
8 Aug 74	1316	-	14[a]	-	0.20	-	0.010[a]
20 Aug 74	1328	-	16	-	0.040	-	<0.020
23 Aug 74	1331	-	23.5	-	0.100[a]	-	<0.020
3 Sept 74	1342	-	16	-	-	-	<0.010
6 Sept 74	1345	-	16	-	-	-	-
17 Apr 75	1568	19.2	-	(0.0127)	-	(.0014)	-
7 Oct 75	1741	20.6	-	(0.0181)	-	-	-
8 Apr 76	1925	18.5	-	(0.0208)	-	(.0007)	-
30 Sept 76	2100	-	-	(0.0263)	-	(.0007)	-
21 Apr 77	2303	18.5	-	(0.0272)	-	.0014	-
13 Oct 77	2479	21.4	-	0.023	-	<0.001	-
19 Apr 78	2666	19.9	-	(0.0272)	-	(.0007)	-
25 Sept 78	2825	23.1	-	(0.0261)	-	(.0007)	-
26 Apr 79	3038	-	-	(0.0362)	-	(.0014)	-
30 Oct 79	3225	21.8	-	(0.0290)	-	-	-
28 Feb 80	3346	-	-	0.05	-	0.002	-
10 Apr 80	3388	21.5	-	0.05	-	0.001	-
15 Apr 80	3393	-	24.6	-	0.047	-	<0.025
22 Apr 80	3400	-	23.2	-	0.046	-	0.0019
28 May 80	3436	-	22.8	-	0.042	-	0.0025
17 June 80	3456	-	23.3	-	0.048	-	0.0029
18 June 80	3457	21.1	-	0.05	-	0.0025	-
24 June 80	3463	-	20.6	-	0.049	-	0.0013
8 July 80	3477	-	21.6	-	0.046	-	<0.010
28 July 80	3497	-	23.6	-	0.052	-	0.0019
25 Aug 80	3525	-	22.6	-	0.052	-	0.0026
9 Sept 80	3540	-	24.0	-	0.053	-	0.0022
17 Sept 80	3548	-	-	0.03	-	0.001	-
6 Oct 80	3567	22.8	-	(0.0453)	-	(0.0007)	-

Table 3. (cont'd)

Date	Days from 1 Jan 1971	Hardness (mg/L)		Zn (mg/L)		Cu (mg/L)	
		ME	DFO	ME	DFO	ME	DFO
15 Oct 80	3576	-	26.5	-	0.046	-	0.0010
17 Nov 80	3609	-	24.5	-	0.053	-	0.0014
27 Nov 80	3619	-	23.7	-	0.047	-	0.0013
10 Dec 80	3632	-	-	0.032	-	0.002	-
11 Dec 80	3633	-	24.9	-	0.045	-	0.0015
22 Dec 80	3644	-	23.1	-	0.051	-	0.0025
12 Jan 81	3665	-	24.9	-	0.066	-	0.0025
2 Feb 81	3686	-	20.5	-	0.054	-	0.0018
19 Feb 81	3703	-	23.4	0.05	-	0.001	-
19 Feb 81	3703	-	-	-	0.053	-	0.0028
3 Mar 81	3715	-	19.5	-	0.064	-	0.0030
24 Mar 81	3736	-	20.0	-	0.0498	-	0.0031
14 Apr 81	3757	22.9	-	0.04	-	0.002	-
28 Apr 81	3771	-	21.2	-	0.0489	-	0.0029
19 May 81	3792	-	20.6	-	0.0616	-	0.0032
8 June 81	3812	-	19.3	-	0.0477	-	0.0022
24 June 81	3828	-	-	0.026	-	0.001	-
29 June 81	3833	-	20.2	-	0.0463	-	0.0035
17 Aug 81	3882	-	21.2	-	0.0486	-	-
20 Aug 81	3885	-	-	0.04	-	0.001	-
1 Sept 81	3897	-	21.9	-	0.0488	-	0.0014
1 Oct 81	3927	21.9	-	(0.0299)	-	(0.0021)	-
13 Oct 81	3939	-	22.4	-	0.0440	-	0.0014
28 Oct 81	3954	-	21.5	-	0.0384	-	<0.0010
10 Nov 81	3967	-	23.0	-	0.0345	-	0.0022
17 Nov 81	3974	-	-	0.03	-	0.002	-
1 Dec 81	3988	-	20.4	-	0.0378	-	-
13 Jan 82	4031	-	19.7	-	0.0348	-	0.0017

[a]Outliers: contamination suspected; not used.

Table 4. Campbell River water quality. Samples taken above the confluence of the Quinsam River. Several unusually high values (Tables 1, 2, 3) have been omitted as outliers where sample contamination is suspected. Analyses for total zinc and copper (ME samples) have been adjusted to equivalent mg/L dissolved metal estimates (Zn X .906; Cu X .708) (see text). DFO estimates are shown as mg/L extractable Zn and Cu. Sampling period: 1971-Jan 1982.

Measure	Period	Minimum	Mean	Maximum
Water harndess (mg/L as $CaCO_3$)	1971-Jan 1982	14	21.5	26.5
pH (Min Envir)	1971-1981	6.6	7.3	7.8
Zinc (mg/L)	1981-Jan 1982	0.0299	0.0464	0.066
(μg/L)	1971-1980	[a]$Zn = 6.73 + 0.004433D + 0.0000019D^2$		
Copper (mg/L)	1980-Jan 1982	0.0007	0.0021	0.0035
	1971-1980	0.0007	-	0.0028
Cadmium (mg/L)	1971-Jan 1982	<0.0005	<0.0005	<0.001

[a]Where D = days from 1 Jan 1971.

Table 5. Short-term LC50 estimates for Zn, generaly arranged in order of increasing water hardness. Median exposure times other than 96-h are indicated in days (ILL-incipient lethal level).

Salmonid species	Size g(cm)	°C	pH	Hardness (mg/L $CaCO_3$)	LC50 (mg/L Zn)	Original reference
coho salmon	0.7	12	6.5	5	0.28	McLeay 1976
rainbow	-	18	6.7	12	0.56	Lloyd 1960
S. gairdneri (alevin)	-	-	-	20	0.090	Garton 1973
sockeye salmon	5-6	12	7.3	20	0.75	Chapman 1978
steelhead (alevin)	-	12.2	7.3-7.5	23	0.815	Chapman 1978
steelhead (swim-up	-	12.2	7.3-7.5	23	0.093	Chapman 1978
steelhead (parr)	-	12.2	7.3-7.5	23	0.136	Chapman 1978
steelhead (smolt)	-	12.2	7.3-7.5	23	>0.651	Chapman 1978
chinook (alevin)	-	12.2	7.3-7.5	23	>0.661	Chapman 1978
chinook (swim-up)	-	12.2	7.3-7.5	23	0.097	Chapman 1978
chinook (parr)	-	12.2	7.3-7.5	23	0.463	Chapman 1978
chinook (smolt)	-	12.2	7.3-7.5	23	0.701	Chapman 1978
rainbow	-	13	6.8	26	0.43	Sinley et al. 1974
rainbow	-	18	6.9	44	0.91(2-d)	Herbert and Shurben 1964
rainbow	28.4	15	7.4	44	0.76	Holcombe and Andrew 1978
rainbow	3.9	15	7.6	47	0.37	Holcombe and Andrew 1978
rainbow	4.9	15	7.6	47	0.52	Holcombe and Andrew 1978
rainbow	-	18	7.1	50	1.2(ILL)	Lloyd 1960
coho salmon	-	11	7.4	94	4.6	Lorz and McPherson 1976
rainbow	3.9	15	7.4	178	2.5(95-h)	Holcombe and Andrew 1978
rainbow	4.9	16	7.2	179	2.96	Holcombe and Andrew 1978
rainbow	28.4	15	7.3	170	1.91	Holcombe and Andrew 1978
S. gairdneri	-	15	7.6	320	3.5(2-d)	Brown et al. 1968
rainbow	-	18	7.7	320	3.8(ILL)	Lloyd 1960
rainbow	-	18	7.8	320	3.9(2-d)	Herbert and Shurben 1964
rainbow	(15)	13	7.8	320	5(2-d)	Herbert and Wakeford 1964
S. gairdneri	73.4	15	7.3-7.9	290	4.6(5-d)	Ball 1967
rainbow	-	16	7.8	333	7.2	Sinley et al. 1974
rainbow	15-17	15.5	7.8	504	4.8	Solbe 1974
Atlantic salmon	9.2	17	7.0-7.4	14	0.420(ILL)	Sprague and Ramsay 1965

Table 6. Chronic and non-lethal response criteria and associated LC50 estimates (mg/L) of response to zinc.

Salmonid species	pH	Hardness (mg/L $CaCO_3$)	LC50 (mg/L Zn)	Comments	Original reference
CHRONIC					
rainbow	6.8	26	0.036-0.071(25-d)	MATC (96-h LC50-0.43)	Sinley et al. 1974
rainbow	6.4	14	1.00(21-d)		Grande 1967
rainbow	7.3	40	0.41(14-d)		Nehring and Goettle 1974
rainbow	7.8	333	0.330-0.640	MATC (96-h LC50-7.21)	Sinley et al. 1974
REPRODUCTION					
brook trout	-	45	0.53-1.36	reduced hatch: egg fragility	Holcombe et al. 1979
GROWTH					
rainbow	-	360	1.12(85-d)		Watson and McKeown 1976
OTHER					
rainbow	-	-	1.43(6-24-h)	decreased arterial O_2, pH	Sellers et al. 1975
coho salmon	-	-	0.5-10.7(1-d)	decreased white cell count	McLeay 1975
rainbow	-	-	0.081(1-d)	hyperglycemia, depletion of	Wagner 1980
rainbow	-	-	0.352(1-5d)	liver glycogen,plasma insulin	Wagner 1980
rainbow	-	-	0.69(1-d)	incr. ventilatory activity	Sellers et al. 1975
Atlantic salmon	-	-	0.01 of LC50	avoidance response	Sprague 1968

Table 7. Responses of representative fish food organisms (LC50, mg/L) to zinc.

Organism	pH	Hardness (mg/L $CaCO_3$)	LC50 (mg/L Zn)	Original reference
Daphnia pulex	-	-	0.50(1-d)	Cairnes et al. 1978
Daphnia pulex	-	-	1.55(1-d)	Cairnes et al. 1978
Daphnia magna	-	-	0.56(1-d)	Cairnes et al. 1978
Daphnia magna	-	-	2.3(1-d)	Cairnes et al. 1978
Daphnia magna	-	-	0.072(64-d)	Anderson 1950
D. sp.	-	6.5	0.5(1-d)	Tabata 1969
D. sp.	-	400	5.2(1-d)	Tabata 1969
D. magna	-	800	<0.65(5-d)	Hutchinson 1933
Ceriodaphna	-	800	>1.3(1-d)	Hutchinson 1933
Gammarus	-	-	<2.0(4-d)	Solbé 1976

Table 8. Short-term LC50 estimates for Cu, generally arranged in order of increasing water hardness (*median exposure times other than 96-h are indicated in days (d) or hours (h)) (ILL-incipient lethal level).

Salmonid species	Size g (cm)	°C	pH	Hardness (mg/L $CaCO_3$)	LC50* (mg/L Cu)	Original reference
rainbow	fry	18-20	-	(soft)	38(1-d)	Turnbull-Kemp 1958
rainbow	-	-	-	-	2(2-h)	Herbert and Van Dyke 1964
rainbow	-	-	-	-	0.2(2-d)	Herbert and Van Dyke 1964
rainbow	-	-	-	-	0.8(2-d)	Herbert et al. 1965
rainbow	-	-	-	-	0.150(2-d)	Cope 1966
rainbow	-	-	-	-	0.4-0.5(2-d)	Brown 1968
chinook	1.35	-	7.2	13	0.010	Chapman and McCredy 1977
rainbow	-	-	-	14	0.037(ILL)	Lloyd and Herbert 1962
Atlantic salmon	(9.2)	17	7.0-7.4	14	0.032(ILL)	Sprague and Ramsay 1965
rainbow	(7.5)	15-17	-	15-20	0.044	Lloyd 1961
rainbow	-	-	-	20	0.050(ILL)	Lloyd and Herbert 1962
rainbow	-	-	-	20	0.100(ILL)	Liepolt and Weber 1958
rainbow	-	15	7.3	21.5	0.033	Howarth and Sprague 1978[2]
rainbow	10	15	8	30	0.06	Howarth and Sprague 1978
chinook	0.3	13-14	6.8-7.2	44	.021-.040(10-d)	Hazel and Meith 1974
chinook	1.35	-	7.6	46	0.025	Chapman and McCredy 1977
sockeye	n.h.[1]	6-9	7.6	83	0.19	Servizi and Martens 1978
sockeye	fry	6-9	7.6	83	0.15	Servize and Martens 1978
sockeye	4.8-5.5	6-9	7.6	83	0.17-0.24	Servizi and Martens 1978
pink salmon	n.h.[1]	6-9	7.6	83	0.14	Servizi and Martens 1978
pink salmon	fry	6-9	7.6	83	0.20	Servizi and Martens 1978
coho salmon	-	10-12	6.8-7.9	93-95	0.074-0.060	Lorz and McPherson 1977
rainbow	3.9	10	7.8	125	0.20	Spear and Anderson 1975
rainbow	29.1	10	7.8	125	0.19	Spear and Anderson 1975
rainbow	176	10	7.8	125	0.21	Spear and Anderson 1975
chinook	1.35	-	8.1	182	0.090	Chapman and McCredy 1977
rainbow	-	-	6.0-6.4	250	0.35(7-d)	Liepolt and Weber 1958
rainbow	(13-15)	17	7.5	250	0.580(3-d)	Brown et al. 1974
rainbow	8.5	17	7.4	240	0.750(2-d)	Brown and Dalton 1970
rainbow	0.36	8-10	7.2	250	0.50(8-d)	Shaw and Brown 1974
rainbow	-	-	-	250	0.85	Min. of Technol. 1973
rainbow	-	-	-	320	0.50(ILL)	Lloyd and Herbert 1962
rainbow	(12.5)	17	7.8	320	0.27(2-d)	Herbert and Van Dyke 1964
rainbow	(12-16)	15	7.4	320	0.89	Calamai and Marchetti 1973
rainbow	(7.6)	16	-	320	1.10(3-d)	Lloyd 1961
chinook salmon	1.35	-	8.5	359	0.125	Chapman and McCredy 1977

[1]n.h. - newly hatched.
[2]Computed for the pH and hardness levels shown.

Table 9. Chronic and non-lethal response criteria and associated LC50 estimates (mg/L) of response to copper.

Salmonid species	pH	Hardness (mg/L $CaCO_3$)	LC50 (mg/L Cu)	Comments	Original reference
CHRONIC					
pink salmon	-	-	0.178	loss of equilibrium, initial mortality	Anderson 1948
pink (young)	7.9	-	0.563	loss of equprilibrium in 19 h	Holland et al. 1960
coho salmon	-	-	1.0	survival, growth, reproductive and feeding responses	Holland et al. 1960
Oncorhynchus	-	-	0.02	sublethal effects on fingerlings	Grande 1967
brook trout	7.5	45	0.0174-0.0095	MATC	McKim and Benoit 1971
rainbow	6.4	14	0.040(21-d)		Grande 1967
coho salmon	7.3-7.4	93-95	.074-.060	96-h LC50	Lorz and McPherson 1977
			.020-.030	osmotic disturbances in blood, suppression of Na^+, K^+-activated ATPASE in part; ceased feeding (6 w to 4 mo)	Lorz and McPherson 1977
			.005-.030	deleterious effect on downstream migration, lowered AtPase activity, reduced subsequent survival in sea water	Lorz and McPherson 1977
REPRODUCTION					
brook trout	-	-	0.010-0.018	reproduction influenced at higher concentrations	NAS/NAE 1974
GROWTH					
chinook	-	-	0.2	inhibition of growth	Hazel and Meith 1970
rainbow	-	360	0.075	threshold for growth inhibition	Lett et al. 1976
rainbow	8	360	0.100	threshold for growth inhibition	Waiwood 1977
rainbow	7.75	100	0.025	threshold for growth inhibition	Waiwood 1977
rainbow	7.5	30	0.015	threshold for growth inhibition	Waiwood 1977
rainbow	6.0	360	0.050	threshold for growth inhibition	Waiwood 1977
rainbow	6.0	100	0.015	threshold for growth inhibition	Waiwood 1977
rainbow	6.0	30	0.010	threshold for growth inhibition	Waiwood 1977
OTHER					
rainbow	-	-	0.056	threshold, avoidance response	Sprague 1968
Atlantic salmon	-	-	0.017-0.021	adult upstream migration delayed	Sprague 1964
Atlantic salmon	-	-	0.0043	threshold, avoidance response	Sprague and Ramsay 1968

Table 10. Responses of representative fish food organisms (LC50, mg/L), or organisms in the food web, to copper.

Organism	pH	Hardness (mg/L $CaCO_3$)	LC50 (mg/L Cu)	Effect	Original reference
Daphnia magna	-	45	0.006	"safe level"	NAS/NAE 1974
Daphnia magna	7.4-8.2	45	0.060(2-d)		Biesinger and Christensen 1972
Daphnia magna	-	soft	0.10	reproduction reduced	NAS/NAE 1974
Daphnia magna	-	soft	0.070	reproduction unaffected	NAS/NAE 1974
Dinoflagellate Amphidinium carberi	-	-	0.075	inhibited growth	Brack et al. 1976
Diatom Nitzschia palea	-	-	0.005	stopped growth	Steeman-Nielsen and Wium-Anderson 1970
Thalassiosira pseudonana	-	-	0.002	50% inhib. of growth	Davey et al. 1973

Table 11. Toxicity of cadmium to some salmonids and fish food organisms.

Organism	Hardness (mg/L $CaCO_3$)	LC50 (mg/L Cd)	Recommendation or effect	Original reference
FISH				
steelhead	23	0.00092	(96-h LC50)	Chapman 1973
rainbow trout	31	-	MATC between 0.0007 and 0.0015 mg/L	Davies 1976
coho salmon	93-95	0.0104	(96-h LC50)	Lorz et al. 1978
salmon eggs, larvae	>100	0.003	recommended max. conc.	NAS/NAE 1974
salmon eggs, larvae	<100	0.0004	recommended max. conc.	NAS/NAE 1974
coho fry	-	0.004-0.001	safe level	McKim and Eaton (unpubl.)
steelhead	-	0.01	(7-d LC50)	Ball 1967
brook trout	-	0.001	(reproductive organs affected	Benoit et al. 1976
brook trout	-	<0.0034	(no effect conc. on spawning)	Benoit et al. 1976
brook trout	-	0.0034	(females died while spawning)	Benoit et al. 1976
brook trout	-	-	alteration of sex hormone metabolism and interference with reproduction when fish exposed to 0.001 mg/L for 21 days	Sangalang and Freeman 1974
FISH FOOD ORGANISMS				
Daphnia magna	-	0.0005	reduced reproduction	Biesinger and Christensen 1971
Daphnia magna	-	0.03	safe level	NAS/NAE 1974
GENERAL				
"Aquatic life"	>100	>0.03	recommended max. conc.	NAS/NAE 1974
"Aquatic life"	<100	0.004	recommended max. conc.	NAS/NAE 1974

Table 12. Toxicity data for fishes exposed to mixtures of Zn, Cu and Cd. (T.U. = toxic units, the fraction of the incipient lethal (ILL) concentration (mg/L).

Organism	Hardness (mg/L $CaCO_3$)	LC50 (mg/L)	Remarks	Original references
chinook	21-27	<0.017 Cu	est. safe levels based on acute tests	Finlayson and Verrue 1980
chinook	21-27	<0.111 Zn	est. safe levels based on acute tests	Finlayson and Verrue 1980
chinook	21-27	<0.011 Cu	threshold for growth effect (max.	Finlayson and Verrue 1980
chinook	21-27	<0.086 Zn	acceptable toxicant concentration)	Finlayson and Verrue 1980
coho	93-95	0.005 Cu	7% of 96-h-LC50-0.071 mg/L interfered with downstream migration	Lorz and McPherson 1977
coho	-	0.010 Cu	interference with downstream migration greater than when either 0.010 Cu or 0.800 Zn presented alone	Lorz and McPherson 1977
Altantic salmon	14	0.032 Cu 0.420 Zn	ILL's when presented (7-d LC50) separately	Sprague and Ramsay 1965
Atlantic salmon	14	0.034 Cu 0.003 Zn	mixture = 1.07 T.U., produced LC50 in 7 days	Sprague and Ramsay 1965
Atlantic salmon	14	0.002 Cu 0.410 Zn	mixture = 1.04 T.U., produced LC50 in 7 days	Sprague and Ramsay 1965
Altantic salmon	14	0.015 Cu 0.230 Zn	mixture = 1.02 T.U., produced LC50 in 7 days	Sprague and Ramsay 1965
Atlantic salmon	18	0.044 Cu 0.580 Zn	ILL's when presented alone	Sprague 1965
			-0.052 of ILL (Cu) produce avoidance -0.092 of ILL (Zn) reactions in -0.021 of ILL (Cu+Zn) juveniles	Sprague 1965
Atlantic salmon	20	-	-0.35-0.43 of ILL - interference with downstream migration of natural migrants, Miramichi	Sprague 1965
fathead minnow	-	>0.4 T.U.*(Cu) >0.4 T.U.*(Zn) >0.4 T.U.*(Cd)	lethal threshold for mixture	Eaton 1973

Table 12. (cont'd)

Organism	Hardness (mg/L $CaCO_3$)	LC50 (mg/L)	Remarks		Original references
rainbow trout (hatch to swim-up)	34-65	0.19 Zn(60-d)	(60-d LC10-0.14 mg/L Zn)	4:1:6	Finlayson and Ashuckian 1979
		0.021 Cu(60-d)	(60-d LC10-0.014 mg/L Cu)	Zn:Cu:Al	
	34-65	0.52 Zn (60-d)	(60-d LC10-0.14 mg/L Zn)	12:1:18	Finlayson and Ashuckian 1979
		<0.010 Cu (60-d)	(60-d LC10-<0.010 Cu)	Zn:Cu:al	
(egg to swim-up)	34-65	0.17 Zn (60-d)	(60-d LC10-0.12 mg/L Zn)	4:1:6	Finlayson and Ashuckian 1979
		0.019 Cu(60-d)	(60-d LC10-0.012 mg/L Cu)	Zn:Cu:Al	
		0.39 Zn (60-d)	(60-d LC10-0.14 mg/L Zn)	12:1:18	
		<0.010 Cu (60-d)	(60-d LC10-<0.010 mg/L Cu)	Zn:Cu:Al	
(swim-up fry)	34-65	0.18 Zn (96-h)	(96-h LC10-0.030 mg/L Zn	12:1:18	Finlayson and Ashuckian 1979
		<0.010 Cu(96-h)	(96-h LC10<0.010 mg/L Cu	Zn:Cu:Al	

Table 13. LC50 values for exposure of chinook and steelhead, and coho salmon, to Zn, Cu or Cd separately (not as mixtures). For the chinook and steelhead, estimates are provided at four stages of juvenile development.

Salmonid species	Metal	Median (h) exposure time	LC50 (mg/L)	°C	pH	Hardness (mg/L $CaCO_3$)	Reference
coho (adults)	Zn	96	0.905	13.7	7.4	25	Chapman and Stevens 1978
coho (adults)	Cu	96	0.046	9.4	7.3	20	Chapman and Stevens 1978
coho (adults)	Cd	215	0.0037	10.0	7.3	22	Chapman and Stevens 1978
steelhead (adults)	Zn	96	0.735*	10.3	7.5	20	Chapman and Stevens 1978
steelhead (adults)	Cu	96	0.035*	9.2	7.6	20	Chapman and Stevens 1978
steelhead (adults)	Cd	408	0.0013*	9.6	7.5	20	Chapman and Stevens 1978
steelhead (adults)	Zn	96	1.755	10.3	7.5	83	Chapman and Stevens 1978
steelhead (adults)	Cu	96	0.057	9.2	7.6	42	Chapman and Stevens 1978
steelhead (adults)	Cd	408	0.0052	9.6	7.5	54	Chapman and Stevens 1978
chinook (alevins)	Cu	96	0.026	12.0	7.1	24	Chapman 1978
		200	0.020				
		200-h LC10	0.015				
chinook (swim-up)	Cu	96	0.019	12.0	7.1	24	Chapman 1978
		200	0.019				
		200-h LC10	0.014				
chinook (parr)	Cu	96	0.038	12.0	7.1	24	Chapman 1978
		200	0.030				
		200-h LC10	0.017				
chinook (smolt)	Cu	96	0.026	12.0	7.1	24	Chapman 1978
		200	0.026				
		200-h LC10	0.018				
steelhead ((alevins)	Cu	96	0.028	12.0	7.1	24	Chapman 1978
		200	0.026				
		200-h LC10	0.009				
steelhead (swim-up)	Cu	96	0.017	12.0	7.1	24	Chapman 1978
		200	0.017				
		200-h LC10	0.009				
steelhead (parr)	Cu	96	0.018	12.0	7.1	24	Chapman 1978
		200	0.015				
		200-h LC10	0.008				
steelhead (smolt)	Cu	96	0.029	12.0	7.1	24	Chapman 1978
		200	0.021				
		200-h LC10	0.007				

Table 13. (cont'd)

Salmonid species	Metal	Median (h) exposure time	LC50 (mg/L)	°C	pH	Hardness (mg/L $CaCO_3$)	Reference
chinook (alevin)	Zn	96	>0.661	12.0	7.1	24	Chapman 1978
		200	>0.661				
		200-h LC10	.364-.661				
chinook (swim-up)	Zn	96	0.097	12.0	7.1	24	Chapman 1978
		200	0.097				
		200-h LC10	0.068				
chinook (parr)	Zn	96	0.463	12.0	7.1	24	Chapman 1978
		200	0.395				
		200-h LC10	0.268				
chinook (smolt)	Zn	96	0.701	12.0	7.1	24	Chapman 1978
		200	0.364				
		200-h LC10	0.170				
steelhead (alevin)	Zn	96	0.815	12.0	7.1	24	Chapman 1978
		200	0.555				
		200-h LC10	0.256				
steelhead (swim-up)	Zn	96	0.093	12.0	7.1	24	Chapman 1978
		200	0.093				
		200-h LC10	0.054				
steelhead (parr)	Zn	96	0.136	12.0	7.1	24	Chapman 1978
		200	0.120				
		200-h LC10	0.061				
steelhead (smolt)	Zn	96	>0.651	12.0	7.1	24	Chapman 1978
		200	0.278				
		200-h LC10	0.084				
chinook (alevin)	Cd	96	>0.026	12.0	7.1	24	Chapman 1978
		200	>0.026				
		200-h LC10	0.018-0.026				
chinook (swim-up)	Cd	96	0.0018	12.0	7.1	24	Chapman 1978
		200	0.0016				
		200-h LC10	0.0012				
chinook (parr)	Cd	96	0.0035	12.0	7.1	24	Chapman 1978
		200	0.020				
		200-h LC10	0.0013				

Table 13. (cont'd)

Salmonid species	Metal	Median (h) exposure time	LC50 (mg/L)	°C	pH	Hardness (mg/L $CaCO_3$)	Reference
chinook (smolt)	Cd	96	>0.0029	12.0	7.1	24	Chapman 1978
		200	0.0023				
		200-h LC10	0.0015				
steelhead (alevins)	Cd	96	>0.027	12.0	7.1	24	Chapman 1978
		200	>0.027				
		200-h LC10	>0.006				
steelhead (swim-up)	Cd	96	0.0013	12.0	7.1	24	Chapman 1978
		200	0.0013				
		200-h LC10	0.0010				
steelhead (parr)	Cd	96	0.0010	12.0	7.1	24	Chapman 1978
		200	0.0009				
		200-h LC10	0.0007				
steelhead (smolt)	Cd	96	>0.0029	12.0	7.1	24	Chapman 1978
		200	0.0016				
		200-h LC10	0.0008				

*Adjusted to values for a hardness of 20 mg/L (as $CaCO_3$).

Table 14. Consolidation of data used to estimate Applicaton Factors for estimating the toxicity of Zn, Cu and Cd to chinook and coho salmon, and S. gairdneri.

Species	Metal	Water hardness (mg/L as $CaCO_3$)	Acute test (96-h LC50) (mg/L)	MATC or close equivalent (mg/L)	Remarks	Reference
chinook	Zn	23	0.097	-	swim-up fry	Chapman 1978
		23	-	0.068	200-h LC10 (swim-up fry)	Chapman 1978
		21-27	-	<0.086	threshold, growth inhibition	Finlayson and Verrue 1980
S. gairdneri	Zn	23	0.093	-	swim-up	Chapman 1978
		23	-	0.054	200-h LC10 (swim-up fry)	Chapman 1978
		26	-	0.036	lower limit, MATC	Sinley et al. 1974
		20	0.090	-	alevins	Garton 1973
coho	Zn	25	0.905	-	adults	Chapman and Stevens 1978
		93-95[1]	1.719[1]	-	migrant smolts	Lorz and McPherson 1977
		5[1]	0.735[1]	-	fry	McLeary 1976
		93-95	-	0.800	slight effect on migration success	Lorz and McPherson 1977
chinook	Cu	24	0.019	-	swim-up fry	Chapman 1978
		24	-	0.014	200-h LC10 (swim-up fry)	Chapman 1978
		21-27	-	<0.011	threshold, growth inhibition	Finlayson and Verrue 1980
S. gairdneri	Cu	24	0.017	-	swim-up fry	Chapman 1978
		24	-	0.007	200-h LC10 (smolts)	Chapman 1978
		21.5[1]	0.033[1]		juveniles	Howarth and Sprague 1978
		15-20	0.044	-	-	Lloyd 1961
		30	-	0.015	threshold, growth effect	Waiwood 1977
coho	Cu	93-95	0.060	-	most sensitive downstream migrants	Lorz and McPherson 1977
		20	0.046	-	adults	Chapman and Stevens 1978
		93-95		0.005	interference with migration	Lorz and McPherson 1977
chinook	Cd	24	0.0018	-	swim-up fry	Chapman 1978
		24	-	0.0012	200-h LC10 (swim-up fry)	Chapman 1978
S. gairdneri	Cd	23	0.00092	-	-	Chapman 1973
		24	0.0013	-	swim-up	Chapman 1978
		31	-	0.0007	lower limit, MATC	Davies 1976
		24	-	0.0007	200-h LC10 (parr)	Chapman 1978
coho	Cd	93-95	0.0104	-	yearlings in Jan	Lorz et al. 1978
		44	-	0.0041	lower limit, MATC	Eaton et al. 1978
		22	-	0.0037	215-h LC50, adults	Chapman and Stevens 1978

[1]Original values adusted to a water hardness of 21.5 (Spear 1981).

Table 15. Estimates of Application Factors obtained from averaged values (mg/L) for (MATC or close equivalent)/(96-h LC50 ratios) from Table 14.

Species	Metal	96-h LC50 (mg/L)	MATC or close equivalent (mg/L)	Application Factor (ratio)
chinook	Zn	0.097	0.077	0.79
S. gairdneri	Zn	0.092	0.045	0.49
coho	Zn	1.120	0.800[1]	0.71
chinook	Cu	0.019	0.0125	0.66
S. gairdneri	Cu	0.031	0.011	0.36
coho	Cu	0.053	0.005	0.09
chinook	Cd	0.0018	0.0012	0.67
S. gairdneri	Cd	0.0011	0.0007	0.64
coho	Cd	0.0104	0.0039	0.38

[1]The MATC for zinc for coho salmon seems high. However it is based on the available data and is accepted here until further evidence is available by which it might be reassessed.

Table 16. Predicted effects of Zn, Cu and Cd (toxic units) on naive chinook, S. gairdneri, and coho salmon juveniles based on maximum, average, and minimum concentrations of the metals in the Campbell River(1971-Jan 1982). The water is predicted to be toxic if $(C_{Zn}/L_{Zn}) + (C_{Cu}/L_{Cu}) \geq 1$, under which conditions C_{Cd}/L_{Cd} is added.

Species	TU at	Zn + Cu + Cd levels, Campbell River, 1971-1982		
		Min (mg/L)	Mean (mg/L)	Max (mg/L)
chinook		0.44	0.79	1.44
S. gairdneri		0.73	1.22	1.78
coho		0.18	0.48	0.78

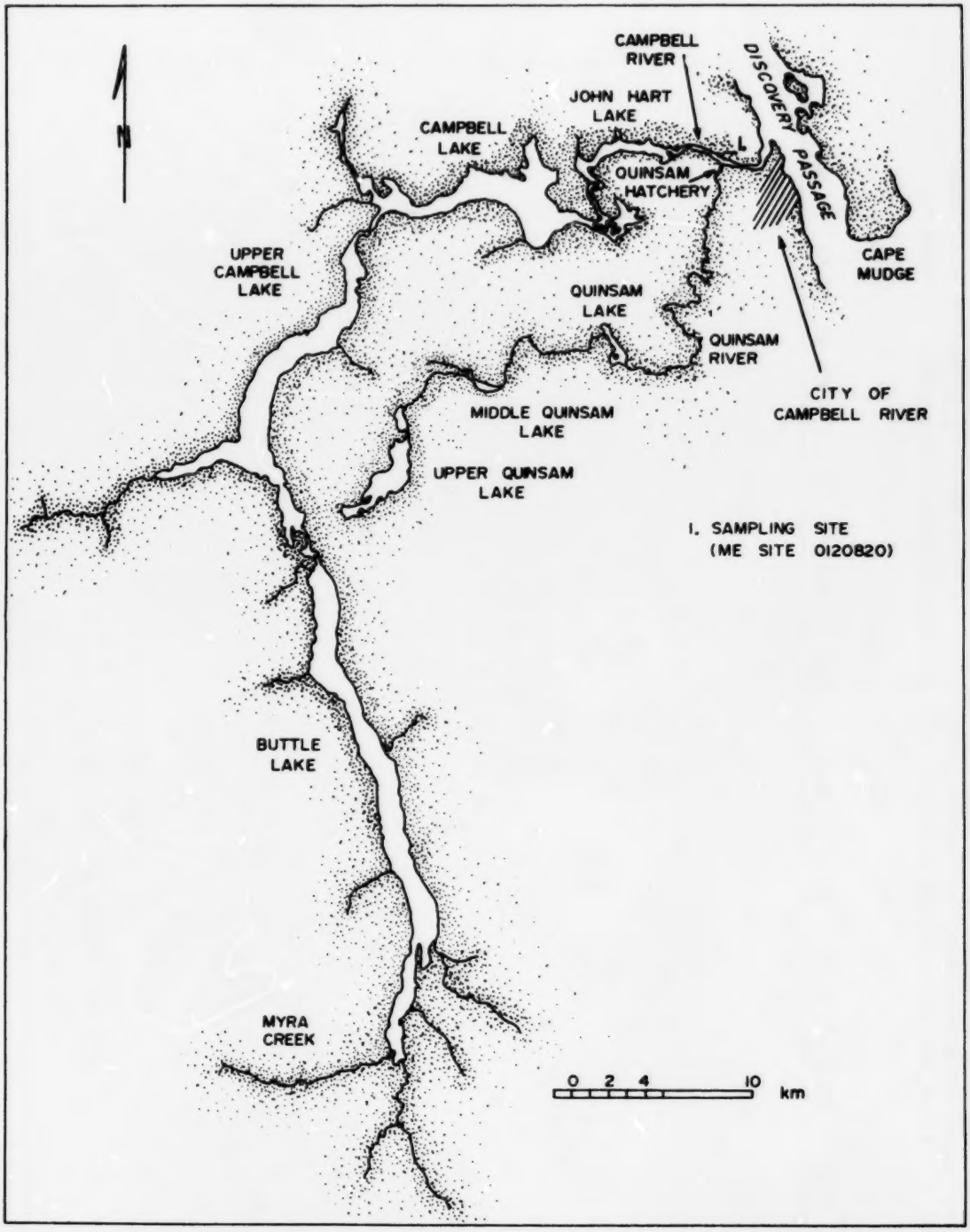

Fig. 1. Sketch of the Campbell River headwater lakes and, below John Hart Lake, the lower Campbell River, its main tributary the Quinsam River, and the Quinsam Hatchery.

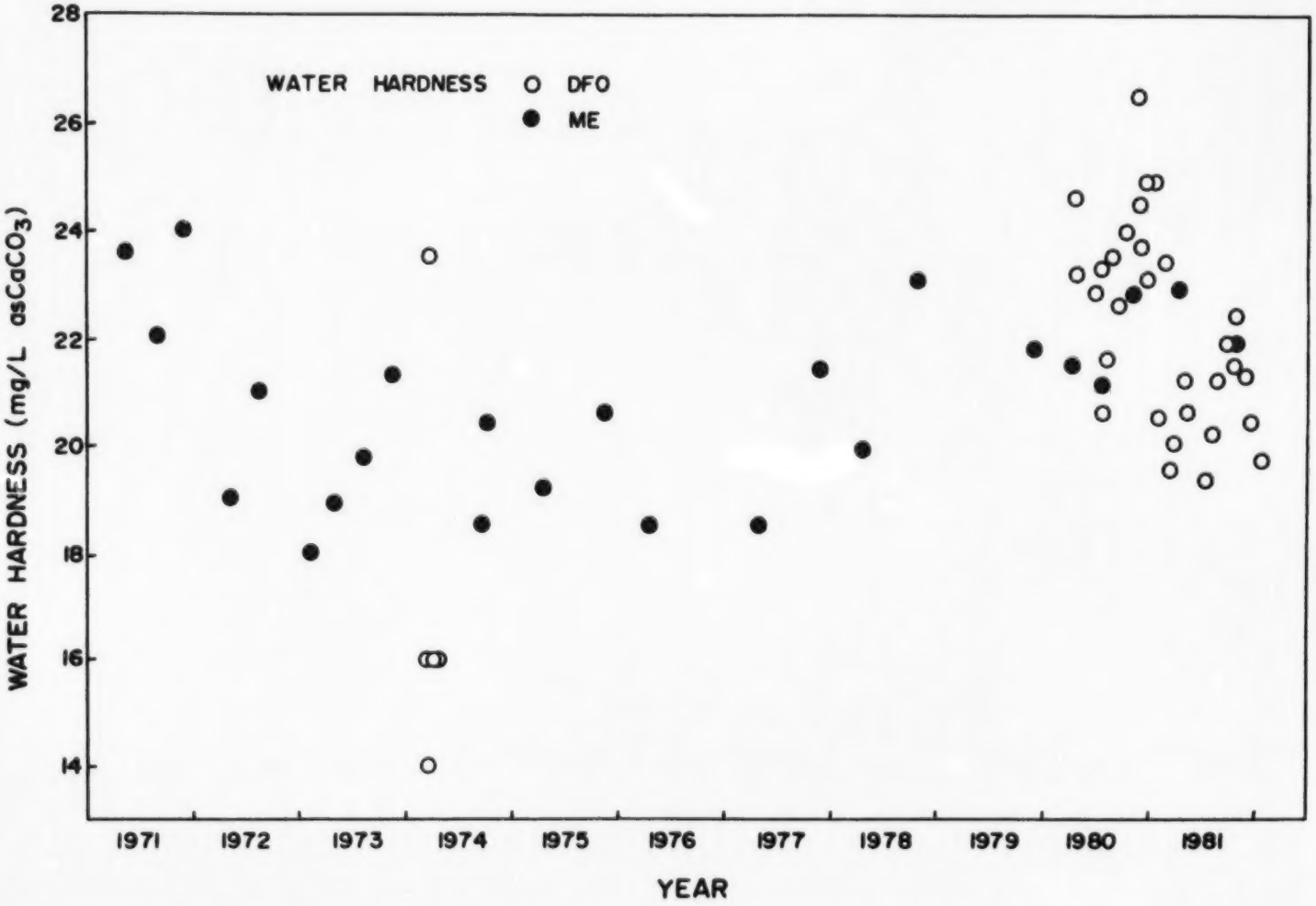

Fig. 2. Water hardness (as mg/L $CaCO_3$) in the Campbell River, 1971-1982. Sources of data: ME--B.C. Ministry of Environment; DFO--Federal Department of Fisheries and Oceans.

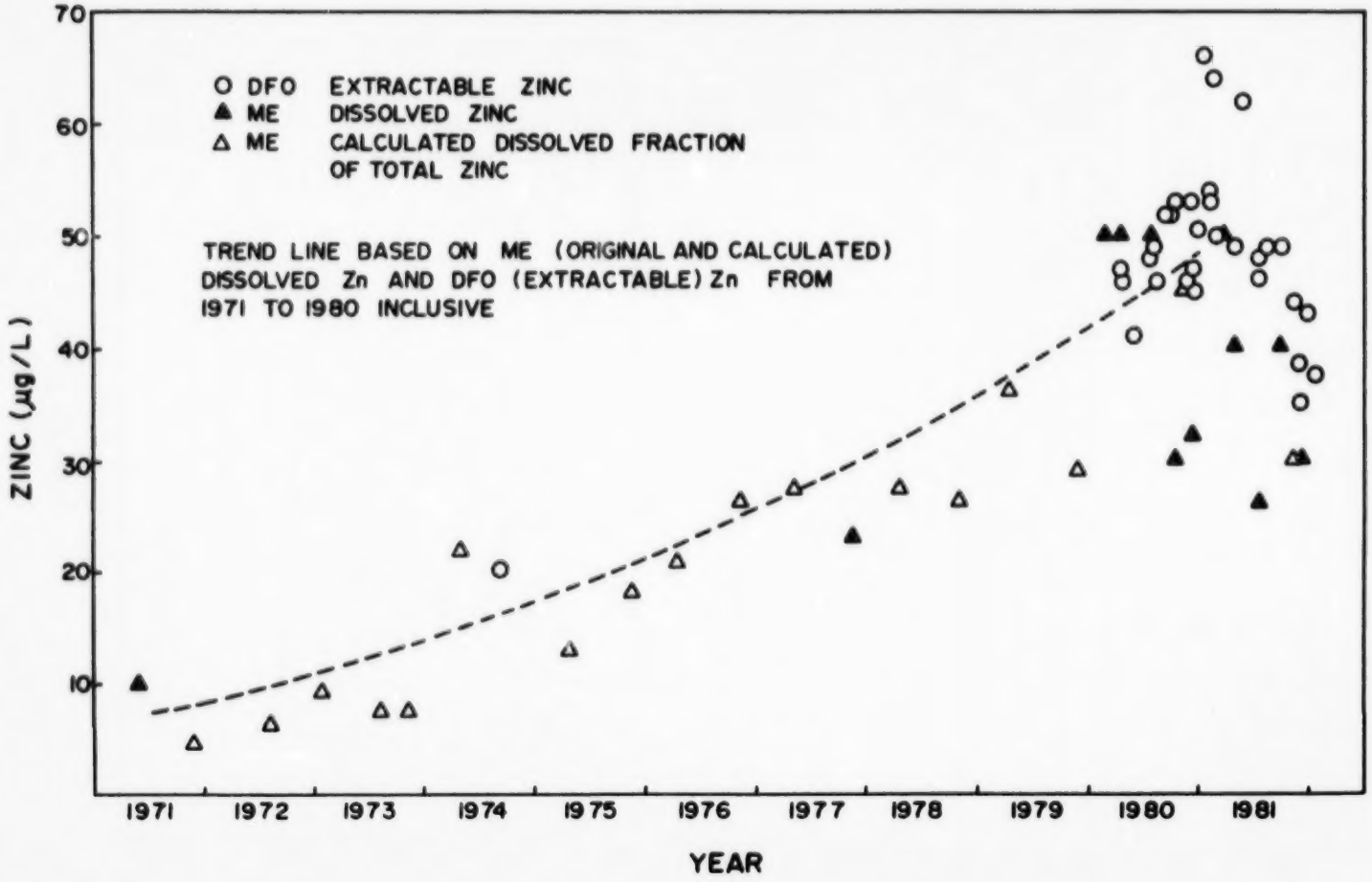

Fig. 3. Zinc in the Campbell River, 1971-1982. Sources of data: ME--B.C. Ministry of Environment; DFO--Federal Department of Fisheries and Oceans. ME data: dissolved zinc and estimates of dissolved zinc obtained from analyses for total zinc. DFO: extractable zinc. All analyses in µg/L.

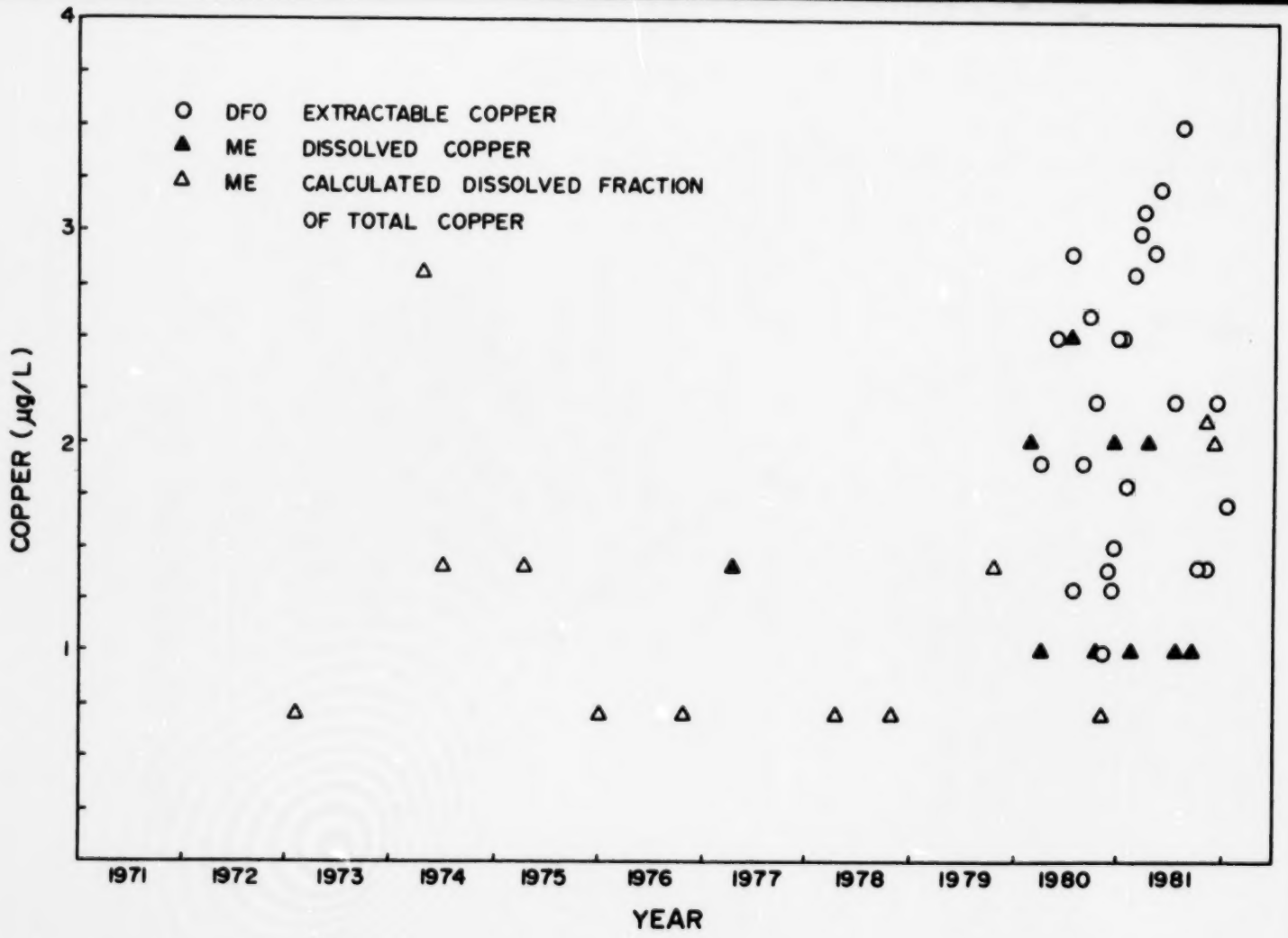

Fig. 4. Copper in the Campbell River, 1971-1982. Sources of data, as in Fig. 3. ME data: dissolved copper and estimates of dissolved copper obtained from analyses for total copper. DFO: extractable copper.

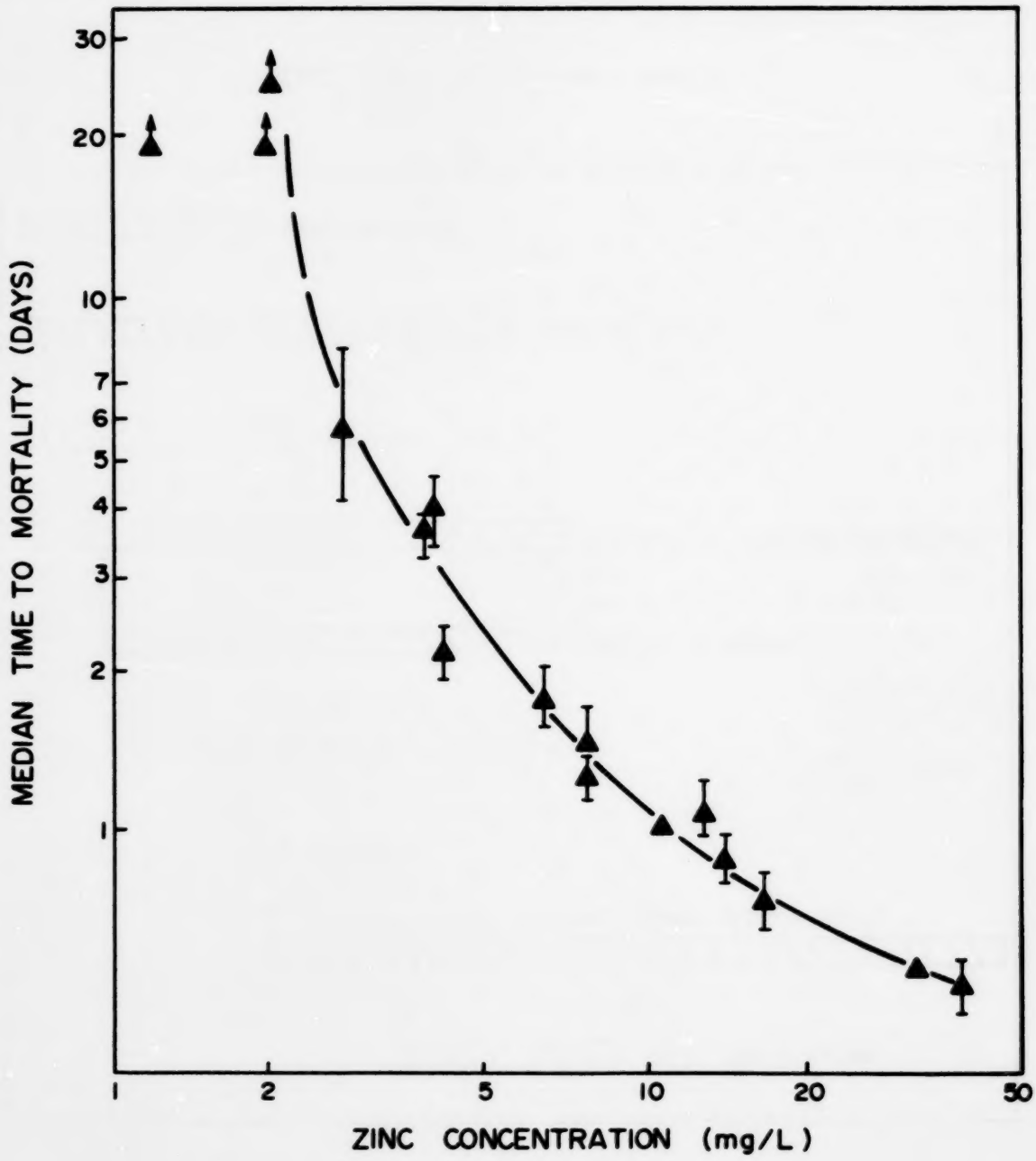

Fig. 5. Representative time-mortality curve showing the time to 50% mortality in relation to various concentrations of zinc to which the fish are exposed. An incipient lethal level would be suggested near 2 mg/L (reproduced with permission from Solbé and Flook 1975).

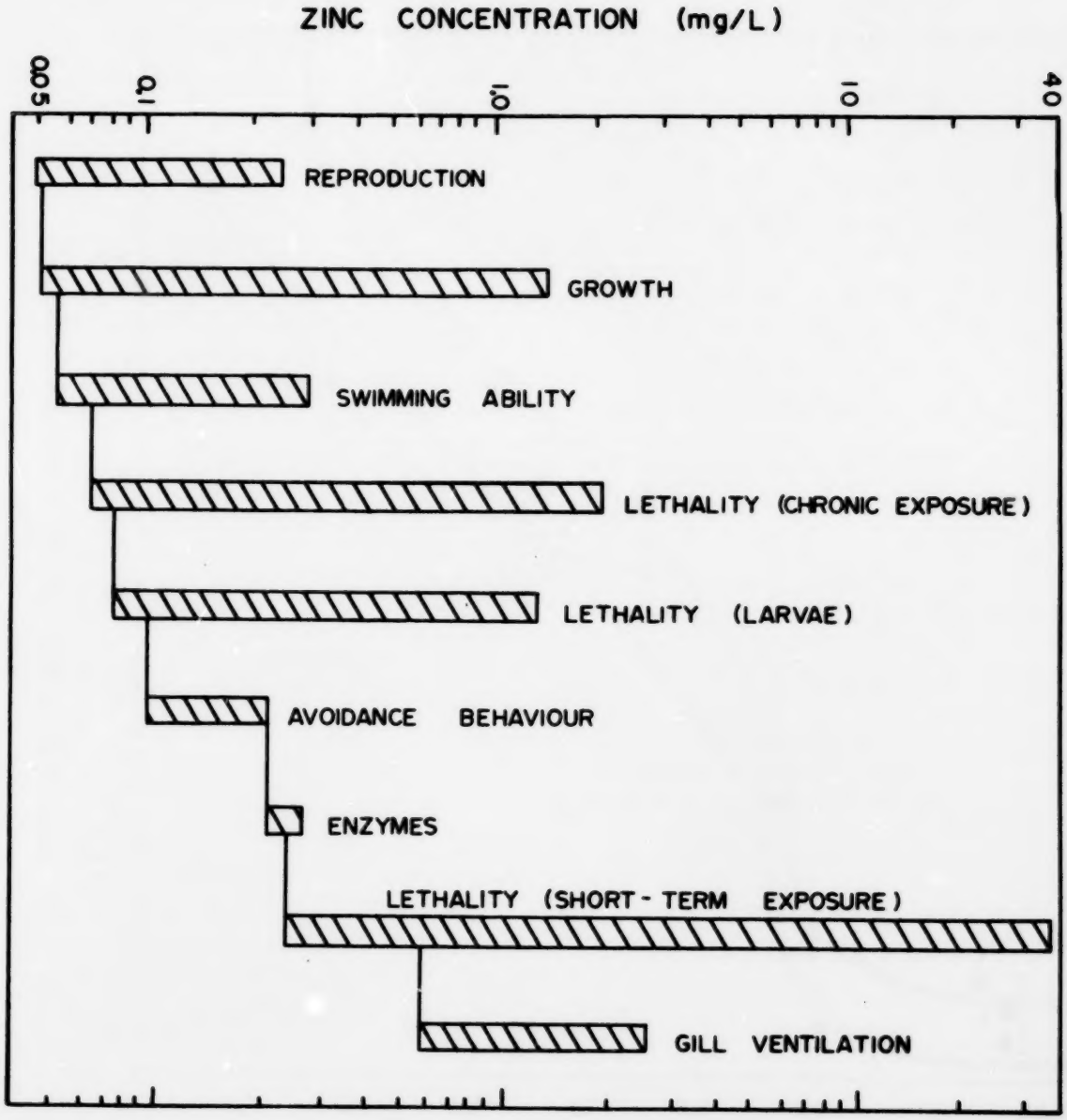

Fig. 6. Comparison of the ranges and lower limits of threshold concentrations known to elicit various lethal and sublethal responses in freshwater fishes (reproduced with permission from NRCC 1981).

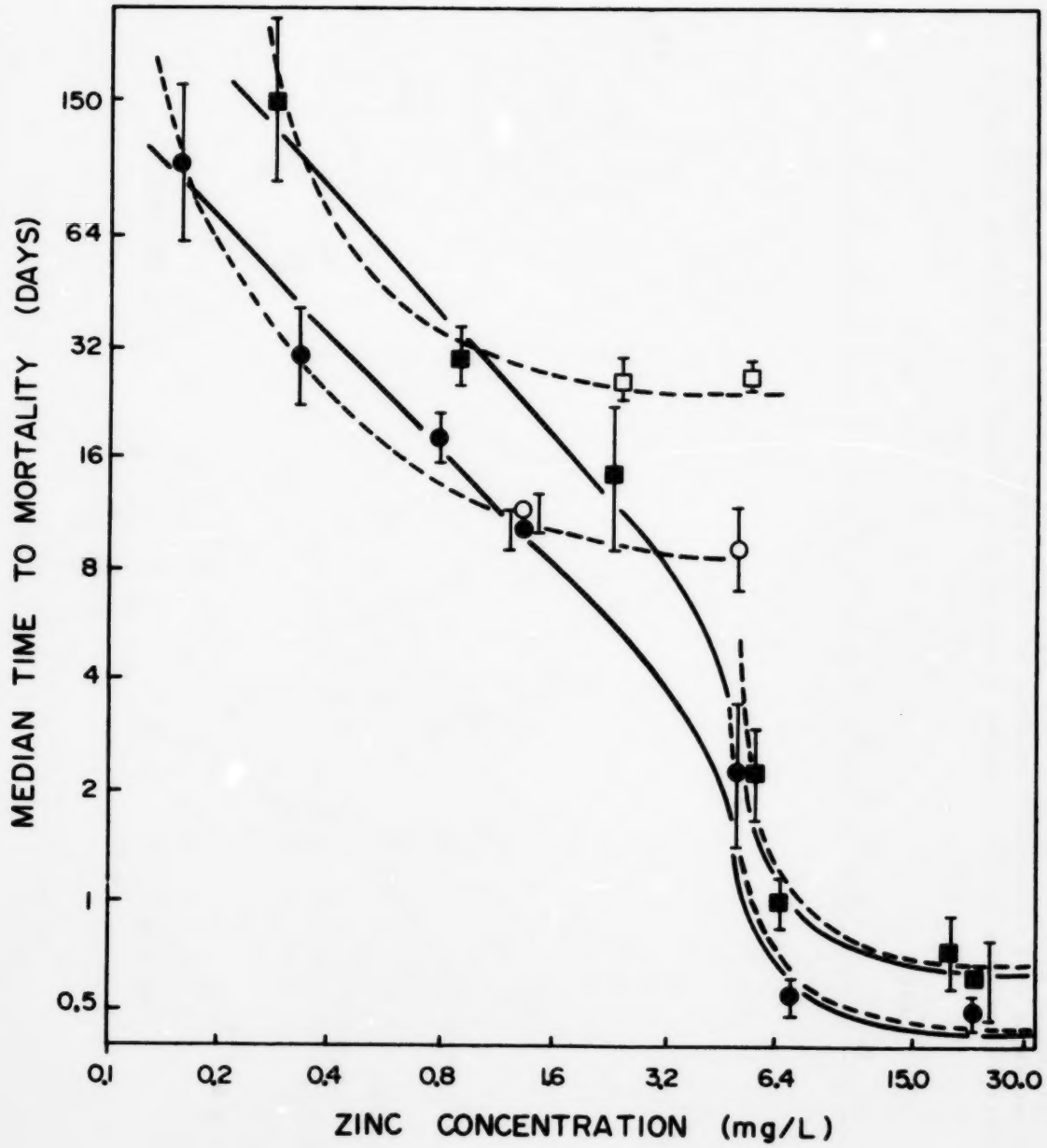

Fig. 7. Time-mortality curve showing two modes of toxic action, the first (acute) with an asymptote near 3-6 mg/L Zn and the second (chronic) below 0.2 mg/L (reproduced with permission from Bengtsson 1974).

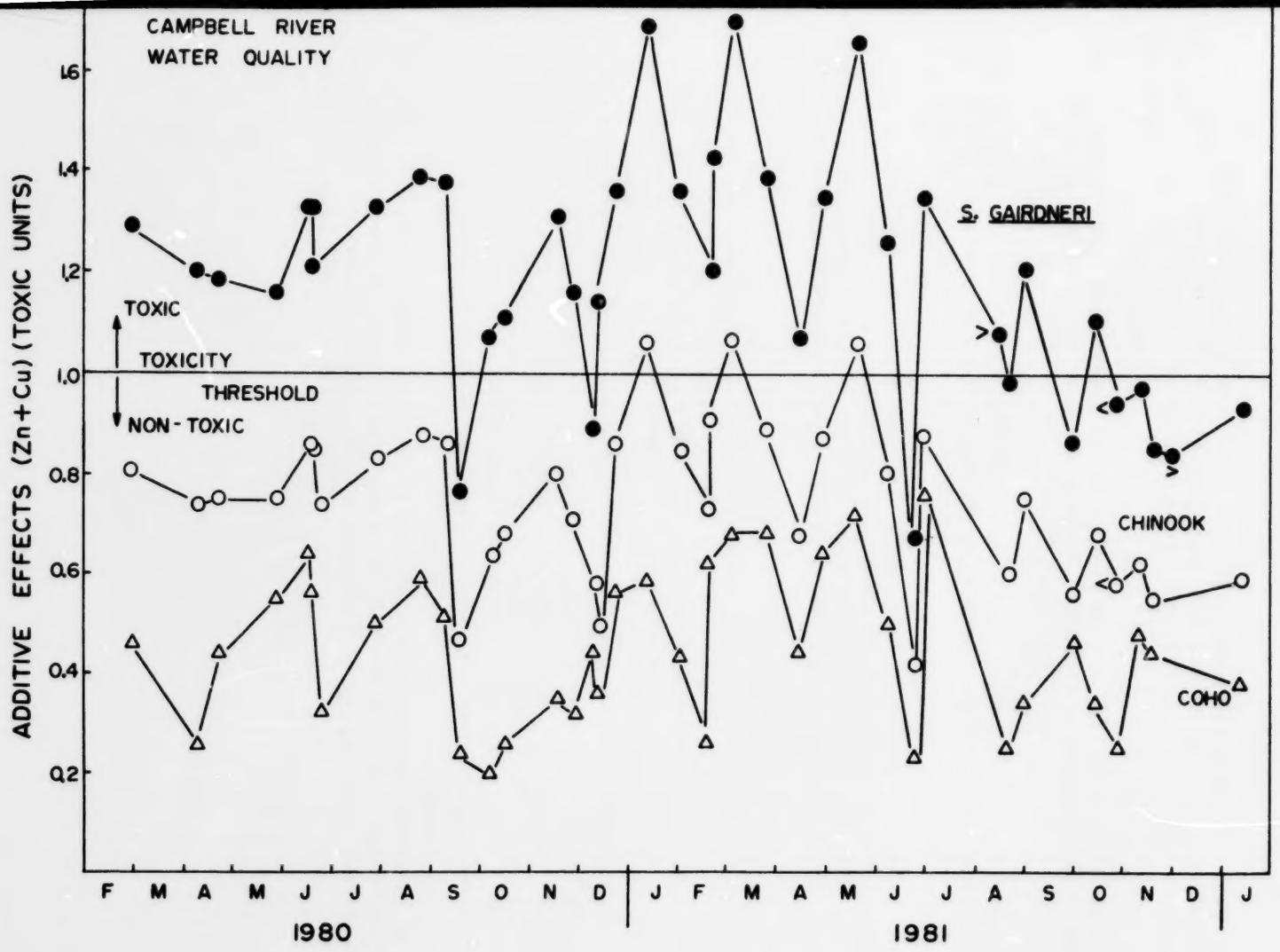

Fig. 8. Additive toxicity of zinc + copper (toxic units), showing the predicted toxicity of Campbell River water to S. gairdneri, chinook and coho salmon for the period 1980-1981 inclusive. Estimates based on analyses from Table 3. Toxic threshold: $C_{ZN}/L_{Cu}+C_{Cu}/L_{Cu} = 1$. Note that water samples are predicted to have been toxic (TU >1) as follows: S. gairdneri--30 of 40; chinook--3 of 38; coho--0 of 38.

8. APPENDIX

When this report was first drafted in February 1982, a number of observations could be made regarding trends and levels of zinc and copper concentration in Campbell River water over the period of 1971-1981. These included,

(1) a continuing upward trend in Zn concentrations, at least until early 1981, followed by an apparent gradual reduction in concentrations during the remainder of 1981.

(2) an apparent but poorly defined trend toward higher Cu concentrations until about mid-1981, followed by an apparent reduction in concentrations during the latter part of the year.

Further samples of Campbell River water have been collected and analyzed in the interim period (Appendix Table 1). Average zinc and copper levels in the Campbell River, in 4-mo intervals between January 1981 and April 1982, and predicted toxic effects (toxic units) for S. gairdneri are as follows.

Period	Average levels (mg/L)		Toxic units			Toxic?
	Zn	Cu	Zn	Cu	Zn+Cu	
Jan-April 1981	.0532	.0024	1.18	0.22	1.40	Y
May-Aug 1981	.0450	.0022	1.00	0.20	1.20	Y
Sept-Dec 1981	.0376	.0018	0.84	0.16	1.00	Y
Jan-April 1982	.0354	.0014	0.79	0.13	0.91	N

In the total period there appears to have been a consistent reduction in both Zn and Cu concentrations. In addition, in terms of predicted toxicity the Campbell River appears to have become non-toxic to S. gairdneri during the latter interval (January-April 1982).

Appendix Table 1. Zinc and copper levels (mg/L) in Campbell River water, January-April 1982 (continuation of Table 3, Fig. 8). The sample for 23 February 82 was analyzed for dissolved metals (ME); the remainder represent analyses for extractable metals (DFO). DFO data: average of duplicate samples. Toxic Units (Zn+Cu) estimated for *S. gairdneri*.

Sample date	Water hardness (mg/L as $CaCO_3$)	Analysis (mg/L)		Equiv. toxic units			Toxic?
		Zn	Cu	Zn	Cu	Zn+Cu	
13 Jan 82	19.7	.0348	.0017	0.77	0.15	0.93	N
4 Feb 82	20.2	.0360	.0015	0.80	0.14	0.94	N
22 Feb 82	19.8	.0380	.0020	0.84	0.18	1.03	Y
23 Feb 82	21.0	.0300	.0010	0.67	0.09	0.76	N
15 Mar 82	20.4	.0385	<.0010	0.86	-	0.86	N
8 April 82	20.3	.0350	.0010	0.78	0.09	0.87	N